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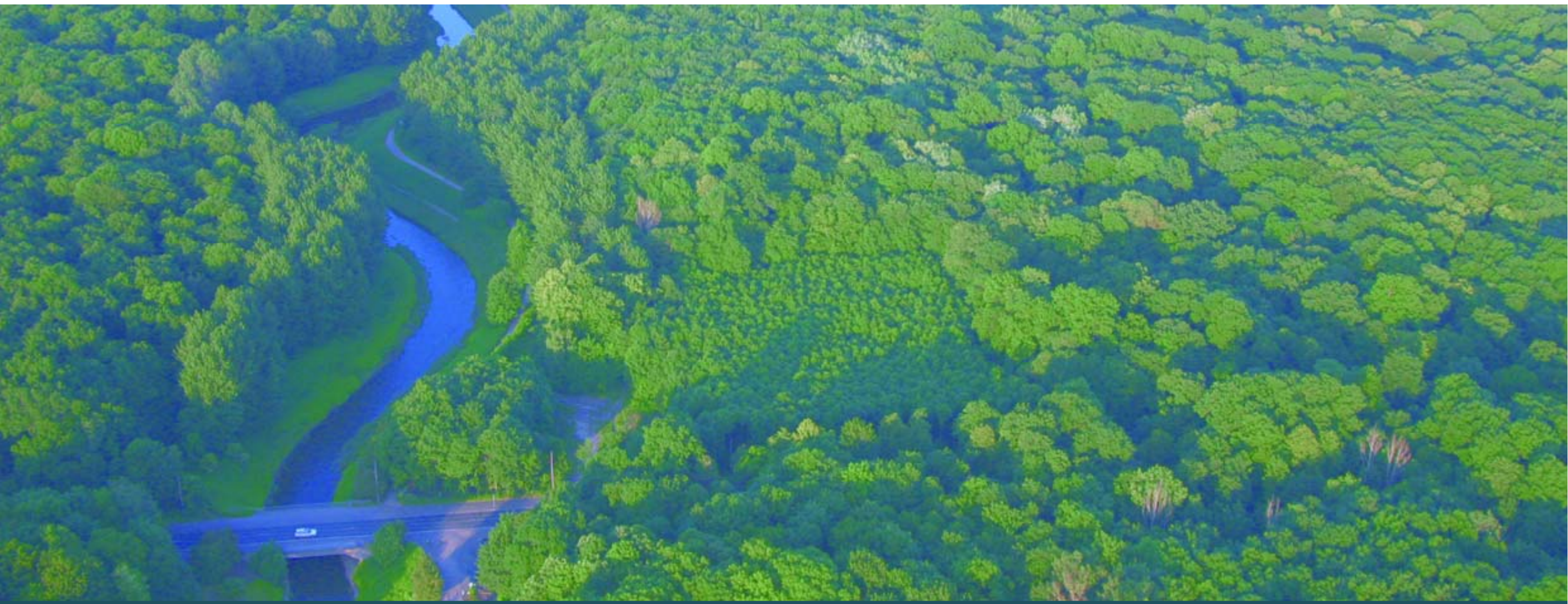

Instrument Mixes for Biodiversity Policies

Irene Ring and Christoph Schröter-Schlaack
(Editors)

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Overview of Contents

Contents

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Leipzig, June 2011 — Irene Ring and Christoph Schröter-Schlaack

Abstract

Biodiversity conservation usually builds on strategies involving a wide range of policy instruments. Within these policy mixes, the use of economic instruments for biodiversity policies and the provision of ecosystem services gains increasing attention, not least in the context of the recent TEEB initiative on The Economics of Ecosystems and Biodiversity. However, there are still many open questions regarding the combination of several instruments in a policy mix. For example, what is the role of economic instruments vis-à-vis regulatory approaches in biodiversity policies? How can the various instruments be assessed in their contribution to conservation objectives, cost-effectiveness, social and distributional impacts as well as institutional requirements, when the focus is on assessing policy mixes rather than single instruments?

Thus far, economic analysis of policy instruments has focused on the assessment of single instruments. The sparse literature available on policy mixes mostly deals with pollution and emissions' related policies rather than with biodiversity conservation and the provision of ecosystem services. Building on first results of the EU-funded project POLICYMIX, this report outlines the challenges involved in assessing the role of economic instruments in policy mixes for biodiversity conservation and ecosystem services provision. We aim to clarify the term policy mix in this application field and provide justifications for using policy mixes instead of focusing on single instruments.

Against this background, we then review important instrument categories for biodiversity conservation and the provision of forest ecosystem services. Instrument types to be covered include regulation and spatial planning, offsets, habitat banking and trading schemes, ecological fiscal transfers, payments for environmental services (PES), forest certification, and – due to its relevance for forest conservation – REDD (Reducing Emissions from Deforestation and forest Degradation) as an international PES scheme.

Finally, a synthesis chapter presents our aggregated findings on assessing policy mixes as well as the role of instruments in a policy mix. We shortly summarise results as presented by the individual reviews and elaborate major characteristics of each instrument or instrument category as regards their roles in a policy mix. In practice, most single instruments do not exist alone; they are implemented in a policy mix context. Some instruments complement each other, whereas others reduce effectiveness and/or efficiency. Therefore, the role of each of the instruments needs to be specified as a basis for further instrument design and impact evaluation.

We elaborate the theoretical interdependencies between different policy instruments and suggest a three-step framework for assessing instruments in policy mixes for biodiversity conservation and ecosystem governance. The first step includes an identification of challenges and context of the relevant conservation problem or case to be analysed. The assessment of policy instruments in a policy mix can further be divided into two steps: 1) what is the specific role of the relevant instrument in the mix in terms of synergies, conflicts or temporal sequencing with other instruments? 2) what is the additional value of the relevant instrument in the policy mix and how can this value be increased or even maximised? With the latter question, more traditional criteria for designing and evaluating policies come into play: one instrument may increase conservation effectiveness, another save costs, yet another contributes to acceptability through more distributive fairness, and finally, some may be required due to legal and institutional requirements in a certain socio-cultural setting.

Introduction

Irene Ring

In most countries, the conservation and sustainable use of biodiversity builds on strategies involving a wide range of policy instruments. Within these policy mixes, the use of economic instruments for biodiversity policies and the provision of ecosystem services gains increasing attention (e.g., McNeely, 1988; Bräuer et al., 2006; EEA, 2006; Ring et al., 2010a)[1]. Especially in the context of the recent international initiative on The Economics of Ecosystems and Biodiversity (TEEB), economic approaches to biodiversity conservation and ecosystem governance gained momentum (TEEB, 2010a, 2010b, 2010c, 2011). However, there are still many open questions regarding the combination of several instruments in a policy mix. For example, what is the role of economic instruments vis-à-vis regulatory approaches in biodiversity policies? How can the various instruments be assessed in their contribution to conservation objectives, cost-effectiveness, social and distributional impacts as well as institutional requirements, when the focus is on assessing policy mixes rather than single instruments?

Thus far, economic analysis of policy instruments has mostly focused on the assessment of single instruments. The sparse literature available on policy mixes deals more often and in a more detailed manner with pollution and emissions' related policies (e.g., Gunningham and Grabosky, 1998; Sorrell, 2003; Sterner, 2003; OECD, 2007; Lindhjem et al., 2009, Lehmann, 2010), rather than with biodiversity conservation, although there are a few notable exceptions (e.g., Young et al., 1996; Gunningham and Young, 1997; OECD, 1999; Doremus, 2003). Due to the comparative novelty of the concept of ecosystem services, assessing relevant policy mixes and the role of instruments in policy mixes for the sustained provision of ecosystem services is even less addressed in the literature. The term 'ecosystem services' was already coined by Ehrlich and Ehrlich (1981), but got more widespread attention not until the late 1990s with the publications of Daily (1997) and Costanza et al. (1997) (de Groot et al., 2010). The ecosystem services concept was then strongly promoted by the Millennium Ecosystem Assessment (MA, 2005a), and this global assessment also included a working group on policy responses (MA, 2005b). However, the MA approach on policy responses was still rather broad and generic. It did not cover a systematic assessment of different instruments in a policy mix.

This report is a result of the EU-funded project POLICYMIX, work package 2 "Review of policy instruments and their roles in a policy mix". It outlines the challenges involved in assessing the role of economic instruments in policy mixes for biodiversity conservation and ecosystem services provision. For this purpose, we develop a pragmatic working definition of the term "policy mix" and apply it to our study focus, the conservation and sustainable use of biodiversity and the sustained provision of forest ecosystem services. We further provide justifications from different disciplinary perspectives for using policy mixes in this application field, rather than building on single instrument approaches.

We shortly present evaluation criteria for instrument choice and single policy instrument analysis that are vast and well established in the literature (e.g., Michaelis, 1996; OECD, 1997; Sterner, 2003). We move on with reviewing and summarising available frameworks for policy mix analysis, with a focus on those that have been presented in the context of environmental policies (Gunningham and Grabosky, 1998; Gunningham and Sinclair, 1999; Sorrell, 2003; OECD, 2007), or even biodiversity conservation policies (Young et al., 1996; Gunningham and Young, 1997).

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[1] *References of the introduction are included in the references section of the following chapter.*

Building on international experience and literature, we then work ourselves towards a POLICYMIX approach for assessing instruments in policy mixes. We first select and then review important individual instruments or instrument categories for biodiversity conservation policies and policies relevant to the sustained provision of forest ecosystem services. While covering regulatory and information-based approaches, we present a larger range of economic instruments. Among the latter are especially innovative approaches to providing economic incentives that have discussed or employed more recently, such as payments for environmental services (PES), ecological fiscal transfers and mitigation banking for biodiversity conservation. We chose from the available tool box a number of instruments that together address both private (PES) and public actors (fiscal transfers). We also decided to include more instruments that reward behaviours beneficial for conservation (compare TEEB, 2011), rather than instruments penalising negative behaviours (e.g., environmental taxes and permits). However, in the latter category, we included biodiversity offsets and banking approaches.

Hence, instrument types to be covered in this review include direct regulation and spatial planning, trading schemes, offsets and habitat banking, tax reliefs for biodiversity conservation, ecological fiscal transfers, payments for environmental services (PES), forest certification, and – due to its relevance for forest conservation – REDD (Reducing emissions from deforestation and forest degradation) as an international PES scheme. Although we aim to provide a generic and global review of the selected instruments, special emphasis lies on instruments for which there is a potential for mutual learning between experiences in Latin American and Europe.

The chapters reviewing individual instruments start with a concise definition and present key features of the instruments in terms of relevant governmental levels and actors involved in design and implementation, next to describing their primary addressees. We discuss the baseline of the instruments from which to assess any improvements in conservation policies. Next follows a description of the range of application of the relevant instrument. In view of POLICYMIX work packages 3 to 6 and the case studies to be performed later in the project as part of work package 7, the major focus of the instruments' reviews is on synthesising the state-of-the-art and knowledge gaps regarding their effectiveness for biodiversity conservation and the sustained provision of ecosystem services, their cost-effectiveness or other means of economic efficiency, their social and distributional impacts, and their institutional context and legal requirements.

The individual instruments' reviews conclude with assessing the role of each instrument in a policy mix. Here we try to answer questions such as: Does the instrument typically operate independently or within a policy mix? Are there hierarchies between instruments? Which other instruments are usually associated with the instrument in question? Are there instruments that complement each other and thus increase effectiveness or efficiency, or reduce costs of the policy mix? Are there other instruments that overlap or conflict with instrument in question, reducing environmental effectiveness or cost-effectiveness?

Finally, a synthesis chapter presents our aggregated findings on assessing instruments in policy mixes for biodiversity and ecosystem governance. We briefly summarise results as presented by the individual reviews and elaborate major characteristics of each instrument or instrument category vis-à-vis their roles in a policy mix. In practice, most single instruments do not exist in isolation; they are implemented in a policy mix context. Some instruments complement each other, whereas others reduce effectiveness and/or efficiency. Therefore, the role of each of the instruments needs to be specified as a basis for further instrument design and impact evaluation.

We elaborate the theoretical interdependencies between different policy instruments and suggest a three-step framework for assessing instruments in policy mixes for biodiversity conservation and ecosystem governance. The first step includes an identification of challenges and context of the relevant conservation problem or case to be analysed. The assessment of policy instruments in a policy mix can further be divided into two steps: 1) what is the specific role of the relevant instrument in the mix in terms of synergies, conflicts or temporal sequencing with other instruments? 2) what is the additional value of the relevant instrument in the policy mix and how can this value be increased or even maximised? With the latter question, more traditional criteria for designing and evaluating policies come into play: one instrument may increase conservation effectiveness, another save costs, yet another contributes to acceptability through more distributive fairness, and finally, some may be required due to legal and institutional requirements in a certain socio-cultural setting.

Justifying and Assessing Policy Mixes for Biodiversity and Ecosystem Governance

Irene Ring and Christoph Schröter-Schlaack

Summary

Biodiversity conservation usually builds on strategies involving a wide range of policy instruments. Policy mixes are even more relevant in the sustained provision of ecosystem services, because further sector policies come into play, be it in a synergistic way or through adverse effects. This chapter aims to clarify the term policy mix in this application field and provides justifications for using policy mixes rather than single instruments. We outline the challenges associated with assessing the role of instruments for success or malfunctioning of policy mixes. We shortly present evaluation criteria for instrument choice and single policy instrument analysis that are vast and well established in the literature. We move on with reviewing and summarising available frameworks for policy mix analysis, with a focus on those that have been presented in the context of environmental policies, or even biodiversity conservation policies. Building on this review of international experience and literature, we then work ourselves towards a POLICYMIX approach for assessing instruments in policy mixes.

1 What is a "policy mix"?

According to Flanagan et al. (2010), the term 'policy mix' first emerged in the economic policy literature in the 1960s, in the context of the relationship and interaction between fiscal and monetary policy. It remained largely confined to these economic policy debates until the late 1980's/early 1990s, when it gained increasing attention by other public policy areas. The most significant extension of the concept could be noted in the field of environmental policy with early publications by Gawel (1991) and Schwarze (1995) or Young et al. (1996) and Gunningham and Young (1997) for biodiversity conservation policies in Australia. Most substantial recent contributions to the policy mix literature occurred in the field of emission-related air pollution and climate policies, where regulatory approaches in the form of technical standards coincide with various economic instruments such as emissions trading and energy taxes (Sorrell, 2003; Lehmann, 2010). Since then, the concept of policy mixes has been taken up in a number of other fields, for example innovation policies (Flanagan et al., 2010).

Despite the increasing use of the concept, clear definitions are often lacking. This is at least the case for the innovation-related literature, where – despite diffuse use of terminology – Flanagan et al. (2010) tend to find normative assertions connected with policy mixes, involving calls for 'appropriate', 'effective' or 'balanced' policy mixes. Comparative assumptions can be found in the literature on conservation policies.

To provide a more rigorous basis for analysing policy mixes in conservation policies in the context of the POLICYMIX project, the following pragmatic working definition is suggested[2]:

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

We focus our analysis on those policy instruments that *positively influence* biodiversity conservation objectives or help sustaining the provision of ecosystem services. We can then distinguish two pathways of analysis:

1. Ex post analysis: In a specific context and at a certain point in time, a mix of *existing instruments* is usually already present. This existing mix can be assessed with a range of evaluation criteria where different instruments contribute to the success or malfunctioning of the overall policy mix in specific ways. To improve the success of the overall policy mix, the focus of analysis may be on:
    a. *one selected policy instrument* that is *to be assessed* against the background of the other instruments in the policy mix using evaluation criteria for single instrument analysis as well as using criteria for analysing policy mixes, or
    b. the *overall policy mix* looking at complementarities or conflicts between several instruments, building on criteria for assessing policy mixes.
2. Ex ante analysis: A new policy instrument is to be introduced against a background of already existing instruments and both the new and the existing ones form the policy mix. In this case, the focus may be on
    a. assessing the *new instrument* regarding its performance as a single instrument, but also in terms of its additional value or conflict potential for the overall policy mix; or
    b. the *overall policy mix including the new instrument* is assessed regarding complementarities or conflicts between several other instruments, using criteria for policy mix analysis.

The mix of existing and new policy instruments for biodiversity conservation and sustained ecosystem services provision is not to be confounded with the *institutional context* or setting of the policy mix as defined above. The institutional context involves the basic institutions of a society, consisting in the formal and informal rules that govern society (economic, political, social institutions). These institutions typically involve property rights, markets, or political governance systems. Furthermore, the institutional context also relates to the policy instruments and mixes in sectors other than biodiversity conservation. Other sectoral policies, such as agricultural, forestry, fisheries or infrastructure-related policies may strongly influence conservation objectives, although often in a negative way in the form of adverse incentives (e.g., TEEB, 2011). If relevant, these effects will be covered in terms of the institutional context in the POLICYMIX case studies.

As a common case study focus in POLICYMIX is the analysis of biodiversity conservation and ecosystem services provided by forests, there is a need for further clarification. We need to distinguish between policy instruments aimed at the conservation and sustainable use of forest resources and the sustained provision of forest ecosystem services on the one hand, and policy instruments that constitute adverse

[2] *As this is an early POLICYMIX project report, and the guidelines for analysing instruments in policy mixes in the case studies are developed in parallel in work packages 3-6, we aim to provide here a first pragmatic working definition.*

incentives on the other. Both types of instrument may belong to forest sector policies. However, the latter would be considered as institutional context, whereas the first-mentioned instruments would be evaluated in their role in the conservation policy mix. When screening the range of ecosystem services provided by forest ecosystems, even more sector policies may come into play. Forest ecosystems are considered to be very important for water regulation and drinking water supply, so instruments related to water sector policies become relevant. If cultural services are to be investigated, tourism and recreation-related policies need to be checked for their relevance. This means that for an assessment of instruments in policy mixes for biodiversity conservation and forest ecosystem services provision, the whole range of instruments in relevant policy sectors needs to be considered – and not just conservation and forest policies.

Individual policy instruments, that are combined to form a policy mix, can be classified according to their major characteristics. Depending on authors and their disciplinary backgrounds, various more or less detailed categorisations have been suggested. The following three major instrument categories have been widely used in the literature (e.g., Michaelis, 1996; Gunningham and Young, 1997; Sterner, 2003):

- Regulatory instruments, including permits, standard-setting and zoning or planning, directly control or restrict environmentally damaging activities.
- Economic instruments, such as environmental taxes, charges and fees, put a price on environmentally damaging behaviour, thus internalising negative externalities, whereas payments for environmental services and ecological fiscal transfers reward conservation enhancing behaviour, thereby addressing positive externalities.
- Informational and motivational instruments aim to shift individual or community preference functions towards more conservation and inform or educate people about relationships between their activities and the environment.

In practical politics, several instruments from these categories can often be found in combination. Some instruments may have been introduced on purpose to enhance the outcome of another instrument. For example, informational instruments are often introduced to provide relevant addressees with the knowledge necessary to enhance the outcome of regulatory or economic instruments. In other cases, economic instruments are introduced as compensation for the costs imposed by regulatory instruments, such as making forest conversion illegal. However, some instruments may simply jeopardise the objectives of yet other instruments. In the case of biodiversity conservation, we usually find instruments of all three categories forming a policy mix. Can this be justified? And in which situations may this be counterproductive?

2 Justifying policy mixes in conservation policies

Justifications for using policy mixes in the conservation and sustainable use of biodiversity emphasise the distinctive character of biological diversity as inherently complex and dynamic (OECD, 1999). Box 1 provides a concise overview on the major arguments justifying the use of policy mixes for biodiversity conservation and ecosystem service provision. ‘Biodiversity’ covers all life on the planet, from the genetic diversity amongst individuals, to the millions of species and the complex terrestrial, freshwater and marine ecosystems they form. Biodiversity and ecosystems provide a wealth of ecosystem services to humans, thereby supporting human well-being (MA, 2005a). Policies for biodiversity conservation and the sustainable provision of ecosystem services contrast with the homogeneous characteristics of other environmental problems that may need to address just one single pollutant. This **heterogeneity** caused by the complex adaptive nature **of biological diversity** and ecosystems involves heterogeneous objectives that naturally calls for a range of different instruments capable of addressing the multidimensional aspects of biodiversity loss and ecosystem degradation.

Box 1. Policy challenges for biodiversity conservation and ecosystem services provision

Heterogeneity and multiple objectives: 'biodiversity' covers all life on the planet, from the genetic level over millions of species to terrestrial, freshwater and marine habitats and ecosystems.

Irreversibility: beyond certain thresholds or 'tipping points', impacts may be irreversible and cause species extinction or ecosystem collapse; appropriate policies adhere to the precautionary principle.

Information gaps: the inherent complexity of ecosystems requires policy decisions under uncertainty; adaptive management approaches.

Mix of values: biodiversity and ecosystems provide a variety of use values and non-use values, some of which are tangible and marketable, whereas others are of a public or common good nature.

(Multiple) market failures: both negative and positive externalities need to be addressed through economic instruments, regulations or the creation of new markets through property rights-based solutions.

Mix of pressures: different pressures on biodiversity and ecosystems arise from various economic sectors, calling for different responses.

Absolute increase in pressures in the face of finite natural environment: the major driver of terrestrial biodiversity loss is land-use change, whose major driver is growth in population and per capita consumption.

Policy failures: many activities with pressures on biodiversity are still subsidised, calling for a reform or removal of adverse subsidies, dysfunctional institutions and corruption.

Impact accumulation: small impacts over a long time may create large losses with irreversible outcomes in the long-run, while the costs of prevention have to be incurred in the present.

Intergenerational equity and discounting: the choice of discount rates is of crucial importance in long-term decisions that range beyond conventional economic calculations, such as dealing with the cost and benefits of biodiversity loss or climate change.

Spatial externalities: whereas the benefits of biodiversity conservation mainly accrue at national and global levels, the costs are often borne at local and regional levels; costs are also unequally distributed between economic sectors, and unevenly spread across administrative units.

Multi-level governance: biodiversity policies require appropriate instruments at local, regional, national and international levels.

Multi-actor governance: due to the multi-facetted nature of biodiversity loss and conservation policies, both public and private actors need to be involved, next to the increasing relevance of hybrid organisations crossing the public-private divide.

Sources: Young et al., 1996; Gunningham and Young, 1997; OECD, 1999: 26; OECD, 2007; Ring, 2008; TEEB, 2008 & 2011.

The necessity of multiple instruments to solve an issue at hand has also been put forward in the context of other environmental problems. In the case of "multi-aspect" environmental problems, the Tinbergen Rule suggests a combination of several instruments, because a first-best optimum cannot be reached with any one single instrument (Tinbergen, 1952; OECD, 2007). In relation to biodiversity and ecosystem services, where multiple problems and objectives are present, Gunningham and Young (1997: 286) suggest that “the number of instruments must be sufficient to accommodate each level of biodiversity and the web of institutions acting to conserve it”. Each threat to biodiversity and each objective would require at least one instrument. But which instruments should be in the policy mix? In practice, this question is difficult to answer. Although the Tinbergen Rule is a useful starting point, it cannot be used mechanically. In this context, OECD (2007) recommends undertaking a careful analysis of the case at hand.

Due to the inherent complexity of biodiversity and ecosystems, considerable **information gaps** exist as regards biodiversity itself or our knowledge on the relationship between biodiversity and ecosystem services (Barbier et al., 1994), although this knowledge has recently been increasing (Naeem et al., 2009; TEEB, 2010b; Ring et al., 2010b). This situation requires policy decisions under uncertainty. Uncertainty and ignorance are not so much perceived as informational failures which can be overcome, but as intrinsic elements and partly an indicator of the richness of biodiversity. Furthermore, human impacts may produce changes beyond certain thresholds or ‘tipping points’ that are often unknown, but possibly irreversible, resulting in species extinctions or ecosystem collapse. Thus, ignorance, uncertainties and informational failures are central in a way that successful conservation policies need to account for the precautionary principle, the idea of safe minimum standards, and adaptive management to prevent major irreversibilities (OECD, 1999). Therefore, any attempts to static optimisation have to be taken with caution in this policy field, as there is no single optimal policy instrument in its own to be identified.

Biodiversity and ecosystems provide goods and services with a **mix of values**, some of which are tangible and marketable, whereas others are of a public or common good nature (OECD, 1999: 27f.). Various economic methods exist for estimating the value of goods whose market prices are imperfect reflections of the total value of these goods or where market prices do not exist at all (Bateman et al., 2011)[3]. It is not surprising that the use and non-use values associated with biodiversity and ecosystems will require different policy instruments. Direct use values, especially when reflecting provisioning services, are privately appropriable, and can often be addressed with economic instruments. Indirect use values reflecting regulating services such as water regulation and purification, and non-use values, including existence values and many cultural services provided by ecosystems, are more difficult to address. Although there are many economic instruments that are suitable to promote indirect and non-use values, they may also need more coercive policy instruments such as regulatory approaches. As most biodiversity benefits are associated with both private and public aspects, instruments have to be implemented that capture both aspects. For example, a rare natural monument can be used by eco-tourists, but also contains an existence value beyond the immediate users. Successful conservation policies thus require combinations of instruments. They “have to use instruments which not only protect direct use values through the provision of well-defined property rights, but also its public values, perhaps through additional instruments such as positive incentives or regulations which guarantee the compatibility of use with the conservation of biodiversity” (OECD, 1999: 36).

Due to the public good nature of many aspects of biodiversity, market failures arise and need to be addressed through regulatory and economic instruments or the creation of new markets through property rights-based solutions. The presence of more than one market failure has already been suggested as an economic reason for policy mixes in the context of climate policies (Sorrell and Sijm, 2003), and we can certainly argue for the existence of **multiple market failures** in conservation policies due to the complexity of problems, resources, and actors. Further economic arguments for policy mixes in

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[3] Bateman et al. (2011) present a concise overview on the range of economic valuation methods as applied to ecosystem services, including further references.

the context of climate policies include prohibitively high transaction costs of solutions with single first-best policies (Lehmann, 2010).

Bennear and Stavins (2007: 117) generally argue that “in a second-best world, coordination of policies is required to attain an efficient outcome” and that an efficient coordination of policies also involves a policy mix of multiple instruments under a fairly wide range of settings. Real-world policy-making is usually pursued in the presence of multiple constraints, such as political constraints, market failures, or policy failures, preventing theory-driven first-best solutions. Hence, second-best problems are characterised by multiple constraints, and each of the constraints “causes the general equilibrium system to fail to reach a Pareto optimum.” The authors discuss five different pairs of constraints that all lead to more than one policy instrument needed to address the problems at hand (Bennear and Stavins, 2007: 117): “(1) imperfect property rights and other externalities, (2) multiple externalities, (3) market power and externalities, (4) unobservable behaviour and externalities, and (5) imperfect information and externalities.” Among the exogenous constraints, that are particularly relevant for justifying multiple instruments, are uncertainties, stakeholder support and administrative capacity constraints (Bennear and Stavins, 2007: 121).

Pressures on biodiversity and ecosystems arise from various economic sectors, calling for different responses. There are uncertainties surrounding the pressures on natural resources and the effects on the resources of these pressures (OECD, 1999). Besides, a number of economic activities with negative impacts on biodiversity and ecosystems are still – at times heavily – subsidised. This includes, for example, subsidies to the fisheries or agriculture sectors, but also infrastructure investments. This argument leads us to policy failures with respect to biodiversity conservation: many activities with pressures on biodiversity are still subsidised, calling for a reform or removal of adverse subsidies as one important aspect of the policy mix for biodiversity conservation (Lehman et al., 2011). Especially when designing payments for environmental services in order to reward the benefits of biodiversity conservation it is crucial to remove counterproductive subsidies first.

Small impacts over a long time may **accumulate** and create irreversible outcomes in the long-run, while the costs of prevention have to be incurred in the present (OECD, 1999). Comparing the costs of policy actions today with the future benefits of either forest investment or conservation projects crucially depends on the choice of **discount rates** (TEEB, 2008; Gowdy et al., 2010). Especially when dealing with long-term decisions that range beyond conventional economic calculations, such as dealing with climate change or biodiversity loss, the ethics of discounting significantly influences **intergenerational equity**.

As with temporal issues, **spatial externalities** represent a widely encountered problem in conservation policies (Ring, 2002; Ring, 2008). Whereas the benefits of biodiversity conservation mainly accrue at centralised levels of decision making, such as national and global levels, the costs are often borne at local and regional levels. Furthermore, costs are unequally distributed between economic sectors, with the primary sector (agriculture, forestry, fisheries) being of extraordinary importance for the conservation and sustainable use of biodiversity and natural resources. Conservation costs are also unevenly spread across administrative units, with some municipalities and districts incurring land-use restrictions related to protected areas, whereas others are free to attract businesses and promote economic development (Ring, 2008). These differences in conservation costs and benefits call for compensatory measures, involving economic instruments of various kinds (Perrings and Gadgil, 2003). The choice of instruments for reconciling the local costs and global benefits of biodiversity conservation depends on who bears the costs (public or private actors) and who benefits from conservation (individuals, businesses, the society at large or the global community).

Biodiversity governance thus is a field where **multiple governmental levels** are relevant, ranging from local, regional, and national level to the international level, each of them requiring an appropriate mix of policy instruments. However, the spatial externalities of biodiversity conservation mentioned above also require instruments capable of addressing interactions between different governmental levels, such as intergovernmental fiscal relations and fiscal instruments. Due to the multi-facetted nature of biodiversity loss and conservation policies, a **multitude of actors** is involved or needs to be addressed in policy making. This includes public and private actors, next to an increasing relevance of hybrid organisations crossing the public-private divide, such as NGOs or semiprivate organisations (for example, agencies and research institutes) (Ring, 2008). Therefore, only combinations of instruments formulated with a wide stakeholder involvement and implemented under specifically designed and context-relevant institutions will lead to successful biodiversity policies (OECD, 1999).

The multiple objectives of biodiversity conservation require a mix of regulatory, property-based, price-based, voluntary and motivational instruments, to achieve these objectives most effectively and to target the range of different pressures at any location. Any individual mechanism entails strengths and weaknesses, and an optimal strategy will focus on the most suitable instrument for achieving an objective, while using additional and complementary instruments to compensate for the weaknesses of individual instruments (Gunningham and Young, 1997). The causes of biodiversity loss and the circumstances in which they arise are so complex and various “that no single instrument, and indeed no single mix of instruments, could be successful in addressing all or even most of them” (Gunningham and Young, 1997: 297). In a similar line, Doremus (2003) argues that the uncertainty remaining about the effectiveness and effects on the future of every conservation strategy requires a broad spectrum of conceivable conservation options. A policy portfolio approach combining several measures promises to prove the best choice for biodiversity protection. A mix of strategies and instruments can address multiple goals, benefit from synergies among various strategies, “reduce the risk of failure and gradually reduce the pervasive uncertainty that currently makes conservation choices so difficult” (Doremus, 2003: 226).

Due to the complexity of all these factors, it is difficult if not impossible “to design a single policy instrument that will successfully provide the right incentives for the sustainable use or conservation of the resources by all the relevant actors. Instead, it is often preferable to employ a range of incentive measures in order to address all the pressures and actors and which, through some overlap in the measures, can provide essential backup in case any one measure fails to provide sufficient incentives.” (OECD, 1999: 12). Taking this last quote seriously, biodiversity policies cannot build on single instruments, nor are complementary instruments sufficient. Due to the ignorance and uncertainties surrounding biodiversity policies and the irreversibility associated with biodiversity loss, it is even recommended to consciously create redundancies between policy instruments.

To conclude, the ecological, social, economic, and political circumstances are so vast that there is no generic ‘optimal’ combination of instruments and mechanisms. “It all depends – the optimal combination will change with time and context” (Gunningham and Young, 1997: 297). Notwithstanding the fact that generalisations are hazardous, some authors have developed a range of generic principles for evaluating and designing policy mixes that will be presented in the sections to follow. In this context, one can distinguish between criteria that have traditionally been used in single instrument analysis, but are also relevant to improve policy mixes, and those that have been suggested for policy mix analysis as such. Nevertheless, these frameworks and criteria for policy mix analysis should be read with care, because concrete recommendations always depend on the specific problem and setting. Turner and Opschoor (1994: 35) go as far as to conclude: “The effects of policy instruments depend on the economic context in which they are applied. A priori general rules are inferior to case-by-case analysis.”

3 How to evaluate instruments in policy mixes?

3.1 Instrument choice and evaluation criteria for single policy instruments

Various authors have developed a number of criteria for the design and evaluation of individual policy instruments. Usually, the ultimate goal in this setting is to choose the best or optimal instrument for a certain setting, although “selecting the ‘best’instrument involves art as well as science” (Goulder and Parry, 2008: 152). This is due to the fact that there is no objective procedure for deciding how much weight to give to different competing criteria. Preeminent criteria regarding the optimality or performance level of an instrument include environmental effectiveness and economic efficiency (Turner and Opschoor, 1994; Michaelis, 1996; OECD, 1997; Gunningham, 1998). Environmental effectiveness relates to the environmental impacts or performance of the instrument (OECD, 1997): How much does the instrument contribute to a defined policy objective? What are its effects on environmental quality? In other words, the marginal environmental benefit associated with a given instrument should be as high as possible (OECD, 2007). Economic efficiency relates to the extent to which an instrument enables a more cost-effective achievement of policy objectives. Efficiency includes both static and dynamic aspects. Static aspects cover the level of administrative costs associated with the instrument to achieve a certain policy objective whereas dynamic aspects relate to extent to which instruments induce technological innovation and / or diffusion (Turner and Opschoor, 1994). Static efficiency can be further divided into a cost-benefit criterion (the marginal cost of implementing a given instrument should be less than its marginal benefit) and a cost-effectiveness criterion (the marginal cost of applying a given instrument should be as low as possible) (OECD, 2007). Policy evaluation studies with just a ‘narrow’ economic focus predominantly or even exclusively look at these two optimisation criteria of the instruments’ effectiveness and efficiency.

However, there are further criteria that contribute to the success of policy instruments. For example, in his framework for assessing allocative impacts of instruments in policy mixes, Gawel (1991) mentions the criteria distributive justice, fairness, political and administrative feasibility, and considers these criteria as further ‘optimisation criteria’. Next to optimisation criteria, coherence criteria play an important role in the afore-mentioned framework. An instrument needs to be generally suitable to reach a certain objective, should be coherent with the legal and institutional system, and unambiguous. In contrast to Gawel (1991), Turner and Opschoor (1994: 11) only consider effectiveness and efficiency as optimisation criteria. In their framework for policy analysis, all other criteria fall under a set of so-called ‘concordance’ criteria, influencing the acceptability of the instrument. These latter criteria include 1) the consistency with policy developments, such as deregulation or policy integration; 2) implications for other policy objectives, for example, relating to the distribution of income; or 3) the general acceptability of instruments or their impacts to vested economic and political interests.

Other authors may rename the criteria mentioned above, (slightly) differ in definition and explanation or add further criteria. For example, in the context of biodiversity conservation policies, Gunningham and Young (1997: 252) add the ‘precaution’ criteria, suggesting that an instrument “avoids the chance of serious or irreversible consequences, especially when there is scientific uncertainty about the outcome”. Bagnoli et al. (2008) provide an excellent overview on methodological approaches to analyse equity issues and the distributional effects of biodiversity policies. Although demanding, they recommend combining institutional and procedural approaches to integrate efficiency and equity considerations into biodiversity policies.

For the purpose of further analysis in POLICYMIX, we build on traditional evaluation criteria mentioned in the literature while moving beyond the core criteria of effectiveness and efficiency in economic analyses, and group them into four basic assessment categories:

- Conservation effectiveness: To what extent does the instrument contribute to achieving a conservation objective? What are its impacts in terms of biodiversity conservation or the provision of forest ecosystem services?
- Efficiency (cost-benefit and cost-effectiveness criterion): What are the benefits of the conservation measures achieved by the relevant instruments? What are the costs of policy implementation? Regarding the latter, we have a special look on transaction costs.
- Social impacts and policy legitimacy: What are the instruments' impacts in terms of equity, fairness, and legitimacy?
- Institutional aspects: How do institutions affect the design and implementation of the relevant policy instruments?

3.2 Frameworks for policy mix analysis

When analysing problems of institutional choice, so many complex configurations of variables need to be addressed that Ostrom (1990: 214; 2009) presented these variables in a 'framework' rather than a single model, because one model could not grasp the necessary degree of complexity. The same applies to instrument choice and instrument design in a policy mix. Policy mix analysis can easily become extremely complex. Owing to the impact of local political and cultural traits, it is very difficult to arrive at global policy conclusions (Gunningham and Sinclair, 1999). Nevertheless, there are a few frameworks that have been developed for policy mix analysis. Often, the starting point is a sector-specific analysis. Young et al. (1996) present a framework for designing policy mixes in biodiversity conservation building on Australian experience and context, whereas Gunningham and Sinclair (1998, 1999) build their framework on preceding research in the chemical industry and the agricultural sector (Gunningham and Grabosky, 1998). Sorrell (2003) and contributors analyse interaction in EU climate policy and from there develop a systematic approach to analyse policy interaction that can be applied in other policy areas. OECD (2007) starts with basic concepts for assessing instrument mixes in environmental policy, while providing lessons drawing on case studies for household waste generation, non-point sources of water pollution, residential energy efficiency, regional air pollution, and emissions of mercury to air. Doremus (2003) focuses on biodiversity protection on private lands to recommend a policy portfolio approach whereas Flanagan et al. (2010) focus on policy mixes in innovation policy.

Frameworks for policy mix analysis often build on or even include evaluation and design criteria that have been used for single instrument analysis as presented in the previous section. Most commonly used criteria include environmental effectiveness, cost-effectiveness, distributional impacts, administrative feasibility and institutional factors. However, for policy mix assessment, these evaluation criteria need to be further developed and additional criteria are required. Looking at policy mixes as a whole, the *relationship* or *interaction* between policy instruments becomes a focus of analysis (Gunningham and Sinclair, 1998; Sorrell, 2003; OECD, 2007; Flanagan et al., 2010). Here, the aim is not to identify the *most* effective or *most* efficient instrument compared to another, but to analyse the interaction between two or more instruments under investigation. Authors promoting policy mixes and policy mix analysis put forward that 'single instrument' or 'single strategy' approaches are misguided because all instruments have strengths and weaknesses (Gunningham and Sinclair, 1999). The task is to build on the strengths of individual instruments, while compensating for their weaknesses through additional or complementary instruments.

3.2.1 Instrument combinations building on 'smart regulation'

Gunningham and Sinclair (1998, 1999) developed a principle-based approach to regulatory design and evaluation of environmental policies, strongly suggesting the superiority of instrument mixes over single instrument strategies. When looking at a mix of two instruments belonging to different instrument categories, Gunningham and Sinclair (1998: 423; 1999) distinguish four basic relationships:

- inherently complementary combinations, where two instruments enhance each other's effect,
- inherently counterproductive instrument combinations, where one instrument negates or dilutes the effect of another instrument,
- sequencing instrument combinations, and
- combinations, where the outcome will be context-specific.

Inherently complementary combinations of instruments significantly enhance the outcome. When used together, the instruments increase overall effectiveness or efficiency irrespective of the environmental issue or the political and socio-cultural setting (Gunningham and Sinclair, 1999). The positive interaction between certain types of instruments holds for a wide range of circumstances, therefore policy makers can be confident to use these instrument combinations without fear of doing harm. Gunningham and Sinclair (1999: 55ff.) present a number of combinations that usually work well together. For example, well-designed informational instruments provide additional value in combination with all other instruments.

Some instruments reduce each other in their efficiency and effectiveness. Irrespective of the context, inherently counterproductive or suboptimal instrument combinations negate or dilute the effect of another instrument (Gunningham and Sinclair, 1999: 61ff.). There are a few combinations that are completely incompatible such as a free market environmentalism/property rights approach in combination with a regulatory command-and-control approach. Other combinations presented by the authors are more complex, such as combining command-and-control regulation with economic instruments which target the same aspect of a problem. In some cases with overriding imperatives (e.g., political or cultural constraints), policy makers may consciously decide to accept the diminished outcome and consciously use counterproductive instrument combinations.

Instruments may be incompatible when employed at the same time, but compatible and reinforcing each other when introduced one after the other (Gunningham and Sinclair, 1998: 444ff.). A typical example of time sequencing refers to self-regulation, followed by stricter standards if the first instrument demonstrably fails to meet predetermined performance benchmarks. One would not use both together; direct regulation would only be introduced when economic actors do not comply with their promises. In other cases, outcomes will be principally context-specific, for example, depend on the political and socio-cultural context.

Building on their detailed discussion of a mix of two instruments with regard to their four basic relationships as explained above, Gunningham and Sinclair (1999) conclude with two general points on multi-instrument mixes. First, additional synergies can often be derived from complementary instruments in policy mixes with more than two instruments. Second, they emphasise the sequence in which the individual instruments are introduced in policy mixes as a potentially crucial factor to their success. Although not yet presented in the framework of the above mentioned four abstract categories of instrument combinations, Gunningham and Young (1997) also discuss combinations of two instruments in biodiversity conservation policies and conclude their article with a number of valuable design criteria toward an 'optimal environmental policy' in biodiversity conservation.

The publications on smart regulation in this section also provide a list of design criteria for instrument mixes in environmental policies or biodiversity conservation policies. For biodiversity conservation (Young et al., 1996; Gunningham and Young, 1997), they include among others 'designing for precaution', 'preference for underlying causes', using 'financially attractive instrument mixes', and 'limiting compensation for a transitionary period'. Van Gossum et al. (2009) apply the smart regulation design principles to their analysis of forest expansion policies and sustainable management of forests in Flanders and the Netherlands, showing that 'smarter' instrument design can contribute to a more successful policy.

3.2.2 Specifying types of policy interaction

Drawing closely on smart regulation theory as presented by Gunningham and Grabosky (1998), Sorrell (2003) and co-authors develop a typology of policy interaction as a basis for policy mix analysis in the context of the EU-funded project "Interaction in EU Climate Policy". They distinguish five types of interaction (Sorrell, 2003: 36), but emphasise that two policies may interact in more than one way:

- Direct interaction involving target groups that are directly affected by two policies and these target groups overlap to some extent.
- Indirect interaction relate to overlapping instruments in terms of the target groups addressed: a) a target group directly affected by one policy instrument overlaps with the target group indirectly affected by a second; b) a target group indirectly affected by one policy overlaps with the target group indirectly affected by another policy.
- Operational interaction where two policies operate together.
- Sequencing interaction, where one policy instrument is followed in time by another instrument, and both directly affect the same target group.
- Trading interaction, meaning that two policies are linked by the exchange of an environmental trading commodity.

Each type of interaction may have implications for the effectiveness, efficiency, social impacts or political feasibility of the policy mix. "Hence, the extent to which such interactions can be judged as beneficial, neutral or counterproductive requires a careful examination of the nature and consequences of the interaction and an evaluation of those consequences within a multicriteria framework. This should lead to a judgement as to whether the combination of instruments is useful, redundant or positively harmful." (Sorrell, 2003: 44)

Moving further to analysing policy interaction, three major steps are suggested (Sorrell, 2003: 44):

1. How and why are two policies affecting each other?
2. What are the consequences of this interaction for the target groups, and the organisations involved in implementing the instrument and aiming to achieve the policy objective?
3. Evaluation of the desirability of these consequences against chosen evaluation criteria.

Interaction analysis can focus on existing or proposed instruments, analyse two or more instruments, and finally aims to identify possible conflicts or synergies between these instruments. Systematic interaction analysis requires comparing the scope of the instruments, the nature of the objectives, the timetable of the instruments, the operation of the instruments, and the process of implementation (Sorrell, 2003: 45).

3.2.3 Policy coherence, positive and negative interaction

In their analysis of instrument mixes in environmental policies, OECD (2007: 22) focuses on policy coherence, positive and negative interactions between policy instruments, and highlights the role of the political context. When judging individual instruments in their contribution to a policy mix in a certain case setting, OECD (2007) primarily asks for the *additional contribution* of the relevant instrument in terms of environmental effectiveness and economic efficiency.

Different levels of policy coordination are required to increase the policy coherence of an instrument mix in environmental policies (OECD, 2007). This coordination is needed among environmental policy instruments and between environmental policies and other related policies, and this coordination is necessary to develop reinforcement mechanisms, or to address possible negative interactions. Coordination among environmental policies is needed to avoid leakage, i.e. solve one environmental problem at the expense of another. Other related policies – the institutional context as defined above – may jeopardise the success of environmental policy instruments, as is the case with adverse subsidies in the fisheries or agriculture sector.

Positive interaction among instruments relates to five different aspects (OECD, 2007: 25ff.):

1. **Providing information:** Information-based instruments such as certification and labelling are often combined with instruments that more directly target the environmental externality (e.g. tax or direct regulation). These combinations can make both instruments more effective, but this depends on the environmental issues at hand, the products to be labelled, and the type of instrument, to which an eco-label is being attached. “Mutual reinforcement among instruments is most likely to be important where there are significant *private benefits* associated with a change in behaviour (e.g. buying labelled products), and the existence of these private benefits will vary with the environmental issue being addressed.” (OECD, 2007: 25). Labelling will most likely also have a larger impact in terms of effectiveness if combined with an economic instrument, where a degree of choice is left to the addressees. Combined with regulation and standards, the impact of labelling may be more difficult to see.
2. **Stimulating innovation:** Supporting research and development activities for technical innovation may lead to positive externalities, but these subsidies need to be properly targeted and designed to avoid a decrease in effectiveness and economic efficiency of the policy mix. In general, financial support needs to provide an incentive for innovations with a positive return for society as a whole. However, there is usually a trade-off between proper targeting of financial support programmes and the additional administrative costs entailed with better targeted measures.
3. **Addressing “split incentives”:** A combination of instruments is useful when market failures exist due to differences between property owners and property users related to a land holding or specific resources. If a land user has to bear the costs of a measure while the benefits accrue to the landowner or vice versa, there is a need for more than one instrument in order to address both stakeholder groups.
4. **Limiting monitoring, enforcement and administrative costs:** Sometimes, more instruments reduce overall costs of compliance monitoring, as was the case, for example, with comparably easy quota systems related to animal numbers in combination with accounting systems for nutrient balances in the Netherlands (OECD, 2007).
5. **Reducing compliance-cost uncertainty:** This argument is put forward for combinations of quantity-based instruments (e.g. quota-based tradable permit systems) with price-based instruments (e.g. taxes). For any relevant activities that are not covered by permits, the economic agent or firm would pay a tax or “fee”. This would limit compliance-cost uncertainty on the side of the regulator (OECD, 2007; OECD, 2008).

According to OECD (2007), major negative interactions in policy mixes relate to different approaches at different governmental levels and to redundancies. Especially in the European Union, EU legislation requires close coordination with domestic instrument design in member states. Furthermore, multi-level governance is crucial both for centrally organised and federal systems, as policy coordination between national, State, Provinces or Länder, and local governmental levels highly contributes to the successful problem-solving. Depending on the situation, overlap between instruments can constitute a problem and increase cost, or dilute the effects of one of the instruments. However, as mentioned above, other authors have even suggested and asked for redundancies in the context of biodiversity policies (Gunningham and Young, 1997). Therefore, this aspect has to be carefully analysed in the context of a specific case setting.

The political context can influence the policy instruments available to a policy mix in different countries. For example, *constitutional settings* differ from country to country. Constitutions define which governmental levels have jurisdiction over which measures and they specify the issues that need to be regulated by laws, thereby leaving the remaining issues to be handled by other decision procedures (e.g. administrative decisions) (OECD, 2007). Theoretically optimal or (more realistically) improved instrument mixes may be difficult to implement due to "political realities", reflecting long-standing political compromises, and where *social policies* and concerns often are core to these constraints. In this context, OECD (2007: 28) recommends to address environmental externalities primarily through environmental policy instruments, whereas social concerns should be primarily addressed through social policies to avoid reducing the effectiveness and efficiency of environmental policies. Finally, OECD (2007) mentions a "*status-quo bias*" among policy-makers that prevents innovative or radical changes in environmental policies despite possible benefits. Main lessons from OECD (2007) are summarised in OECD (2008).

3.2.4 From policies to governance frameworks

Drawing on the strengths and weaknesses of smart regulation theory that, in the words of Van Gossum et al. (2010: 245), is characterised by "almost infinite 'smart' regulatory options", the concept of "regulatory arrangements" has been put forward. The latter approach constrains the manifold options of smart regulation theory by "the national policy style; adverse effects of policy arrangements of adjoining policies; the structure of the policy arrangement of the investigated policy and competence dependencies or other institutions." (Van Gossum et al., 2010: 245). In their approach they highlight the relevance of policy learning, the institutional context and governance capacity to introduce certain instruments in a specific context.

Other authors have also emphasised the multiplicity of instruments operating in a policy setting. Bressers and O'Toole (2005) stress the social and political context of applying instruments and the networked character of implementation contexts. The authors deem it crucial, "that instruments be analysed in their mutually reinforcing – or sometimes impeding – combinations" (Bressers and O'Toole, 2005: 134). For effective governance, policy analysis needs to go beyond a perspective focusing on separate and isolated instruments. The authors distinguish several joint forms of influence, or 'confluence', in policy mixes (Bressers and O'Toole, 2005: 137ff.):

- Increased intensity of policy intervention, meaning that "more than one instrument can be targeted simultaneously at the same group to intensify a policy intervention";
- Integration of multiple instruments into one interactive process between government and target groups, for example to address several actors in the same process;
- Instruments and actions at different levels of governance;

- Competition and cooperation between different but interdependent policy fields;
- Mutual strengthening or weakening of the effects of interventions at different points of action in the broader social and ecological system.

Bressers and Kuks (2003) characterise instrument mixes for the purpose to get assistance in analysing policy formulation, implementation and contributing to the effectiveness of instruments in view of the target groups. The introduce 'five multiplicity aspects of governance': Multiple levels of governance, multiple actors in policy networks, a multiplicity of problem definitions and other policy beliefs, multiple other instruments and multiple responsibilities and resources for implementation. Although instrument selection and design are still an important topic in the work of Bressers and co-authors, they clearly move beyond a mere focus on instrument and policy analysis. The social and political context, in which instruments are introduced and implemented, becomes much more important. Policy analysis is shifting towards governance analysis in a multi-level and multi-actor context, or in the authors terms a 'networked context'.

In a similar line, Flanagan et al. (2010) analyse policy mixes in innovation policies in a multi-level and multi-actor context. They distinguish between dimensions and forms of interaction, next to possible sources of tension between instruments in a policy mix (see Figure 1).

Dimensions of interaction	*Forms of interaction*
Policy 'space' Governance Geography Time	Between different instruments targeting the same actor or group (within or across dimensions) Between different instruments targeting different actors/groups involved in the same process (within or across dimensions) Between different instruments targeting different processes in a broader 'system' (within or across dimensions)
	Between nominally 'the same' instruments (across the different dimensions)

Possible sources of tension between instruments in the policy mix
Conflicting rationales Conflicting goals Conflicting implementation approaches

Figure 1. Conceptualising policy mix interactions: dimensions, forms of interaction and potential sources of tension

Source: Flanagan et al. (2010: 26)

The dimensions of policy mix interaction relate to *policy space* as the abstract space in which different policy domains coexist, the *governance level dimension* relating to interactions across different hierarchical levels of governance, the *geographical* dimension, the policy mix interactions in real space, and the *time* dimension. They use Bressers and O'Toole's (2005) framework as their starting point to build a more sophisticated conceptualisation of interactions, ending up with three kinds of policy mix interaction in the case of unambiguously distinct instruments (Flanagan et al., 2010: 24):

- “interactions between instruments targeting the same actor or group of actors,
- interactions between instruments targeting different actors/groups involved in the same process, and
- interactions between instruments targeting points of action which may seem to be far removed but which interact because the processes or actors targeted prove ultimately to be linked by other processes in a broader ‘system’.”

A fourth type of interaction relates to the interaction of ‘the same’ instruments across one or more dimensions, for example between different levels of governance or in time. Finally, Flanagan et al. (2010) include negative interactions into their framework in the form of tensions between instruments in a policy mix. These relate to conflicting rationales, goals and implementation approaches.

3.2.5 Further approaches and concluding remarks

There are a few other authors who envision a range of policy options to support biodiversity protection. These approaches may not yet be considered encompassing frameworks for assessing instruments in policy mixes, as they focus on specific aspects in the overall analysis. Nevertheless, these studies are valuable contributions to developing a framework for assessing instrument mixes in biodiversity policies.

Doremus (2003), for example, suggests a portfolio approach to biodiversity protection on private lands. In her paper, she presents different types of instruments and her favoured metrics for evaluating policy options. Interestingly, the author’s focus is first, on the feasibility of the policy instrument, then on effectiveness, fairness and future effects of the policy instrument. From her analysis of conservation options available, she concludes (Doremus, 2003: 226): “A mix of strategies can address the divergent goals typical of policy decisions, profit from synergies among the various strategies, reduce the risk of failure, and gradually reduce the pervasive uncertainty that currently makes conservation choices so difficult.” Although there is a focus on policy mixes in biodiversity conservation policies, this paper does not develop a framework for policy mix analysis. Likewise, Stoneham et al. (2003) suggest a portfolio of policy mechanisms to achieve the multiple objectives of biodiversity conservation on private lands. Their primary aim is to define an efficient set of policy instruments and they suggest a transaction costs approach to identify an optimal policy portfolio in this setting.

In the light of the more encompassing frameworks as presented above, how do we move forward from here? At this stage, the key lessons may be framed as following:

Policy mix analysis does not primarily ask whether one instrument is more effective or efficient than another, assuming only the more effective instrument should be used. The interesting question for policy mix analysis is on *interaction* between instruments. Are combinations of instruments complementary to each other, are they mutually reinforcing, do they involve conflicts when present at the same time, or are they suitable to be introduced one after the other in a temporal relationship to increase outcome? Furthermore, there may be situations where no general recommendation is possible at all, and the outcome completely depends on the context.

Positive and negative interaction between instruments may be defined differently for biodiversity conservation policies compared to general environmental policies. Regarding the latter field, OECD (2007) mentions overlap of instruments as a potential source of inefficiency and thus includes overlap in the category of ‘negative interactions’. To the contrary, overlap of instruments is even recommended by several authors for biodiversity policies (Gunningham and Young, 1997; OECD, 1999) and thus, subsumed under positive interactions. We argue that overlap or functional redundancy of individual instruments increases the resilience of the policy mix: When there is large environmental heterogeneity and variability as is the case for biodiversity, there will be ignorance about instrument effectiveness (as opposed to measurable uncertainty or risk underlying a portfolio philosophy). In such situations functional redundancy and policy experimentation in adaptive management may be appropriate.[4]

What has been dealt with as the social, political or institutional context in earlier frameworks of policy mix analysis, seems to become a focus of analysis in later frameworks. In recent years, instrument choice and design as well as policy mix analysis has more and more been complemented by *governance* analysis, as the role of the state has continuously changed, and other actors enter stage, among them non-governmental organisations, businesses, or civil society representatives.

4 Towards a POLICYMIX approach to assessing instruments in policy mixes

4.1 Reviewing instruments and their role in a policy mix

This report reviews available literature and recent developments in policy analysis related to relevant single instruments as well as policy mixes. The major objective of our findings is to support the development of the POLICYMIX project's concerted approach to assessing the role of economic instruments in policy mixes for biodiversity conservation and ecosystem service provision. Parallel to the instrument reviews as the primary task of work package 2, guidelines for the assessment of instruments in policy mixes in the POLICYMIX case studies will be developed in work packages 3 to 6. Building on the state-of-the-art and recommendations from this report, these latter work packages develop a modular and stepwise evaluation framework for carrying out policy assessment with available data in a number of European and Latin American case studies (more information on case studies at http://policymix.nina.no). More specifically, the tasks are as follows:

Work package 3 on 'Ecological effectiveness of policy instruments: Gains in biodiversity conservation and ecosystem services provisioning' will scope the availability of data for ecological effectiveness analysis in the case studies. Methods and models for quantifying gains in biodiversity conservation and in ecosystem service provisioning based on biodiversity inventories as well as land-use and land-cover data will be reviewed. Depending on available data, a tiered approach of indicators of biodiversity conservation gains and ecosystem service provisioning will be developed. Based on experiences with biodiversity surrogates and ecosystem service modelling in case studies, a framework and best-practice guidelines to assess synergies and trade-offs in biodiversity and ecosystem service provisioning will be developed.

Work package 4 on 'Economic benefits and costs of economic instruments and their implementation' will conduct a critical review of the biodiversity valuation literature with the aim of developing practical guidelines for biodiversity valuation and the assessment of the economic benefits of economic instruments. Similarly, a cost accounting framework will be developed to improve transparency and consistency in economic policy instrument implementation cost accounting and better integrate the associated transaction costs into decision-making on the ground. The benefit valuation and cost

[4] *This is an approach borrowed from ecology in e.g. the design of agroforestry systems in the face of climate change.*

accounting guidelines will be tested in the case studies, providing the basis for best practice methodological guidelines.

Work package 5 on 'Social impacts of instruments and enhancing policy legitimacy' will review the application of social impact in policy instrument analysis, including both distributional (outcome) and fairness (process) issues, in order to develop a conceptual framework. The framework will be developed in the form of methodological guidelines for analysis of the instrument mixes in the case studies. Based on experiences, best practice guidelines for social impact analysis of policy instruments will be completed.

Finally, work package 6 on 'Institutional and legal options and constraints' will develop a framework for analysing the impacts of institutions on policy mixes. Focus will be both on formal institutions (WTO law, EU state aid law, EU biodiversity law) and informal institutions (practices, traditions, beliefs) and how they enable and constrain the adoption of particular policy instruments and their implementation. Using a common conceptual framework, country specific institutional analyses will then be carried out in the case studies.

To sum up, we build on traditionally used criteria in economic policy analysis, such as effectiveness and cost-effectiveness, to assess individual instruments in comparison to other instruments in policy mixes for biodiversity conservation and forest ecosystem services provision. However, we expand this catalogue to include social and distributional impacts as well as institutional factors, which we consider as essential, complementary criteria to assess the performance of instruments in policy mixes.

Work package 7 on 'Assessment of existing and proposed policy instruments for biodiversity conservation: case studies' will then implement the evaluation frameworks in an integrated manner in seven case studies, assessing the roles of economic instruments in policy mixes at national and local governmental levels. Work package 8 on 'Multi-scale comparative case study analysis and transferability assessment of economic instruments' ensures consistent comparison of the methodological approaches and components developed in work packages 3 to 6 in comparative cross-case analysis at national and local level. Last, but not least work package 9 on 'Methodological synthesis and policy recommendations' will revise guidelines for multi-scale policy mix assessment which were initiated in work package 2, detailed in work packages 3 to 6, and tested in the case studies. Policy mix design and case study transferability will be synthesised together with best-practice recommendations.

4.2 Selecting key policy instruments for biodiversity and ecosystem governance

As stated above, POLICYMIX focuses on the analysis of those instruments that *positively influence* biodiversity conservation objectives or help sustaining the provision of ecosystem services. Regarding the type of ecosystems considered there is a focus on ecosystem services provided by forest ecosystems. In our review of policy instruments, we aim to cover all major categories of policy instruments, i.e. regulatory instruments, economic instruments as well as information-based instruments. There are a number of different instruments within each of these categories that may also be appropriate to be considered for relevant policy mixes due to the multiple actors and governance levels involved in biodiversity conservation and ecosystem services provision. Table 1 provides an overview of the major characteristics of different instruments in terms of the incentives provided, the incentivising actor, the incentivised actor and the relevant conditions.

Table 1. Characteristics of biodiversity conservation instruments

INSTRUMENTS	INCENTIVE	INCENTIVISING ACTOR	INCENTIVISED ACTOR	CONDITION
Regulatory instruments				
Direct regulation and spatial planning	coercion	government	public and private resource user	various behaviours that are generally or in this instance **negative** for the environment
Economic instruments				
Biodiversity offsets and mitigation banking	avoiding fine	government	private resource user	project planned that involves a **negative** environmental impact
Environmental taxes	tax	government	private resource user	various behaviours that are generally or in this instance **negative** for the environment
Tax reliefs	avoiding tax	government	private resource user	various behaviours that are generally or in this instance **positive** for the environment
Ecological fiscal transfers	payment	government	government body negatively affected by regulation	enforcement of regulation or various behaviours that are generally or in this instance **positive** for the environment
Environmental subsidies	payment	government	private resource user	various behaviours that are generally or in this instance **positive** for the environment
Government financed payments for environmental services (PES)	payment, contract	government	private resource user	compliance with terms of contract
Market-based payments for environmental services (PES)	payment, contract	rival resource user	private resource user	compliance with terms of contract
Voluntary and information-based instruments				
Voluntary instruments	prevention of coercive regulation	government (indirectly)	private resource user	compliance with voluntary agreement or pledge
Certification	avoiding regulated loss of access to market or gaining good consumer reputation	government, private market operator, consumers or NGOs	private resource user	compliance with code of conduct, etc.

Source: own compilation by Christian Klassert, UFZ.

Regarding regulatory instruments, we cover direct regulation as well as spatial planning as the major approaches of the so-called 'command-and-control' type of instruments (Schröter-Schlaack and Blumentrath, this report). These instruments often form a basis for the application of economic instruments in biodiversity policies. Information-based instruments belonging to the category of voluntary instruments will also be covered, but for the review, we specifically focus here on forest certification (Kaechele et al., this report), given its relevance for forest conservation and sustaining forest ecosystem services.

As POLICYMIX has a special focus on assessing economic instruments, we purposely select different economic instruments capable of covering a wide range of problems, issues, and actors. We include in our review offsets, habitat banking and tradable permits as economic instruments building on quantity-based approaches. Offsetting negative impacts on nature in the course of infrastructural investments or other projects have a comparably long tradition in some countries (e.g. Germany). In contrast, habitat banking and tradable permits have been suggested and investigated more recently in the context of biodiversity conservation (see Santos et al., this report).

There is a wide range of price-based approaches available for providing economic incentives to both public and private actors. Both well-known and well-treated in the literature are environmental taxes to internalise negative environmental externalities. However, more important for biodiversity and ecosystem governance are economic instruments that reward the benefits provided by both private *and* public actors through payments and markets (TEEB, 2011). Therefore, we chose as relevant topics for the following instrument reviews tax reliefs (Oosterhuis, this report), ecological fiscal transfers (Ring et al., this report), and Payments for Environmental Services (PES) (Porras et al., this report). There is no clear distinction between environmental subsidies and PES; in as far they incentivise positive environmental behaviour, they are subsumed under the PES category. Subsidies in sectors other than biodiversity policies, however, can often act as adverse incentives promoting environmentally harmful activities. If relevant, these effects will be covered in terms of the institutional context in the POLICYMIX case studies. Due to their relevance for forest conservation at the global scale, we include a review on REDD (Reducing Emissions from Deforestation and forest Degradation) and REDD+ as a recent extension to include forest conservation and sustainable management principles. REDD/REDD+ initiatives can be considered as an 'architecture of policy instruments' rather than a single instruments. They are still in development and foresee international payments for environmental services (Chacón-Cascante et al., this report).

To sum up, we will review in this report important instrument categories for biodiversity conservation and the provision of forest ecosystem services with a special focus on economic instruments. Instrument types to be covered include regulation and spatial planning, offsets, habitat banking and trading schemes, tax reliefs, ecological fiscal transfers, payments for environmental services , forest certification, and – due to its relevance for forest conservation – REDD and REDD+ as an international PES scheme.

4.3 The structure of the individual instrument reviews

The instruments or instrument categories to be reviewed in the following chapters build on a common structure: Each chapter starts with a concise definition and presents key features of the instruments in terms of relevant governmental levels and actors involved in design, decision-making and implementation, next to describing their primary addressees. We discuss the baseline of the instruments from which to assess any improvements in conservation policies. Next follows a description of the range of application of the relevant instrument. For example, is it primarily used for conservation purposes or is the major application of the instrument outside environmental policies? If the latter applies, what options exist to include environmental aspects?

To support guideline development in work packages 3 to 6 and in view of the case studies to be performed later in the project, the major focus of the instruments' reviews is on synthesising the state-of-the-art and knowledge gaps regarding

- the effectiveness of the instrument for biodiversity conservation and (forest) ecosystem service provision,
- the cost-effectiveness or other means of economic efficiency,
- the social and distributional impacts, and
- the institutional context (including the legal context).

Finally, the individual instruments' reviews conclude with a preliminary assessment of the role of each instrument in a policy mix. Here we try to answer questions such as: Does the instrument typically operate independently or within a policy mix? Are there hierarchies between instruments? Which other instruments are usually associated with instrument in question? Are there instruments that complement each other and thus increase effectiveness or efficiency of the policy mix? Are there other instruments that overlap or conflict with instrument in question, reducing environmental effectiveness or cost-effectiveness? These insights then form the basis for the final synthesis chapter of this report.

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Direct Regulation for Biodiversity Conservation

Christoph Schröter-Schlaack and Stefan Blumentrath

Summary

Policies subsumed under the term 'direct regulation' have long been and still are the most widely used approaches for environmental protection and biodiversity conservation in particular. However, so far direct regulation has perceived only little attention in economic research on instrument choice. Hence, the purpose of this literature review is threefold: (1) to describe the main features and provide a classification of direct regulation; (2) to survey the available literature regarding the environmental effectiveness, cost-effectiveness, distributional aspects and institutional prerequisites of direct regulation; and (3) to draw conclusions regarding the role of direct regulation within a policy mix for biodiversity conservation and ecosystem service management.

1 Introduction

Measures subsumed under the term 'direct regulation', such as technological and environmental standards and spatial planning, have long been – and still are – the most widely used approaches for environmental protection. Such approaches are used to establish protection objectives (e.g. for species and habitat protection), to reduce pollution (e.g. to air, water and soils), to prevent hazardous events (e.g. in dealing with toxic substances) and to trigger urgent environmental improvements (e.g. by banning certain substances entirely). 'Direct regulation' is especially important for biodiversity conservation (protected areas and species protection) and ecosystem service management (best management practices in agriculture and forestry).

Against this backdrop it is astonishing, that 'direct regulation' has perceived only little attention in economic research on instrument choice. Often it is simply used as a baseline in stylised modelling to demonstrate the superiority of more flexible approaches, such as market-based instruments. Typically, 'direct regulation'-measures are judged as 'traditional' and 'inefficient' policies not able to establish least-cost solutions in environmental management.

Hence, the purpose of this literature review is threefold. Firstly, we describe main features and provide a classification of the measures subsumed under 'direct regulation' in the field of biodiversity conservation (Section 2). Secondly, we've surveyed the available literature regarding the environmental effectiveness, cost-effectiveness, distributional aspects and institutional prerequisites of 'direct regulation' (Section 3). Finally, conclusions regarding the role of 'direct regulation' within a policy mix for biodiversity conservation and ecosystem service management are drawn (Section 4).

2 Definition and key features

2.1 Definition

The term **'direct regulation'** covers a broad range of different instruments and thus making it challenging to establish a comprehensive definition. Typically, it is understood as prohibiting certain activities or prescribing certain actions to be taken by the addressed actors. Very often the term '**command and control**' is used synonymously, revealing the idealised mechanics underlying 'direct regulation' – to command certain behaviour and to control actors for their compliance.

Most economic text books discuss 'direct regulation' against the backdrop of pollution control. Based on the work done there and the classification elaborated, one could distinguish three basic types of regulatory instruments that may affect biodiversity and thus the provision of ecosystem services (amended after Hansjürgens et al., 2011b; Sterner, 2003):

- **Regulation of technology**: involving regulation on resource management and production such as technical standards (e.g. Best Available Technique (BAT) in production), management prescription for good practices (e.g. 'good agricultural practices', 'good forestry practices' or obligations to reforest harvested stands), and restrictions on the use of products (e.g. ban of pesticides, illegally logged timber or non-endemic species in afforestation efforts);
- **Regulation of performance**: involving regulation requesting a certain environmental status such as emission standards or ambient quality standards (e.g. 'good ecological status' of water bodies as requested by the European Water Framework Directive) or the protection of certain species (e.g. by prohibiting or restricting hunting, harvest or trade and by safeguarding habitats);
- **Spatial planning**: involving regulation with different levels of binding force (see Fürst, 2005, 2008a, b). One can distinguish three means with an increasing degree of compulsion: 1. information, 2. balancing and coordination[1], and 3. the setting of standards and norms (e.g. restricting certain land-use types to defined zones). Based on generated or processed information these modes of operation are applied in all phases of the (political) planning process (see figure 1). In doing so, spatial planning aims at guiding future action of the planning agency and addressees while facilitating targeted, concerted action with a long-term perspective.

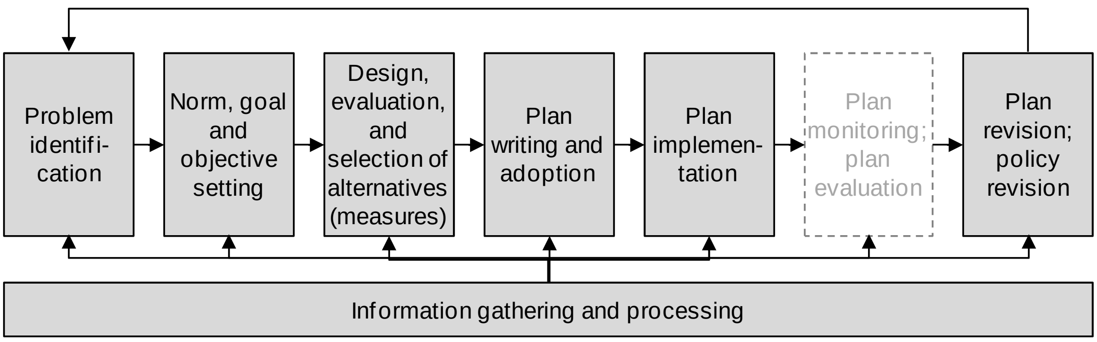

Figure 1. Process of planning, generalised and idealised
Source: based on (Fürst, 2008a: 49)

It is obvious, that the three-tiered classification of 'direct regulation' above is neither exclusive nor comprehensive and available evidence for the performance of regulatory instruments for biodiversity conservation may only occasionally be ordered along these categories. However, as will be shown in the following, it still makes sense to classify approaches regarding their regulatory impact – whether they are requiring (a) a certain technology with a rather undefined environmental performance, (b) a certain environmental performance without detailed prescription on the technology to use or (c) a certain institutionalised method to consider the different economical, ecological or distributional effects of land use related activities.

[1] *In the context of planning, coordination is rather understood as a targeted process of interaction where the participants adapt their ways of thinking and acting to one another than as to unilaterally influence others behaviour (Fürst, 2005).*

2.2 Actors involved

Design and implementation of legal instruments are solely public responsibilities. The different **governmental levels act as 'regulator'** whereas **private actors and subordinated government authorities are 'addressees' of 'direct regulation'**. The regulating authority sets the standard, monitoring and enforcement often resides with some bureaucratic agencies and offices or may even be organised in cooperation with the regulated actors. However, there are also examples of regulations developed by negotiation; either with a single regulated party or with an entire regulated community (see for two examples from the US Coglianese, 1997; Harter, 2000).

In case of **spatial planning**, the decision making processes typically involves **stakeholder participation** (including different public agencies responsible e.g. for transport or nature protection, NGOs, business associations, or affected landowners) (Fürst, 2005: 23; van den Berg, 2005: 76). **Also participation of the general public** has been introduced into many fields of (spatial) planning, inter alia following the Åarhus convention (UNECE, 1998), which was ratified by 40 (primarily European and East Asian) countries till the end of 2009. In line with this, a move towards participatory and community-based approaches can be observed especially in the last two decades. This "can be seen as a backlash against more elitist technocratic, top-down models of policymaking that historically have been prevalent in natural resource management institutions" (Steelman, 2001: 71). These characteristics further separate this type of 'direct regulation' from the other two. Additionally, setting up **private forest management plans** is common practice in the forestry sector of several countries (e.g. Germany or Norway), which makes spatial planning not only a matter for public authorities. Finally, planning is becoming more and more an 'intermediate authority', acting as a kind of interface between economic, political and social steering systems (Fürst, 2005: 23). In practice, planning is developing a new form of (regional) governance, aiming at collecting players from politics, administration, business and NGOs to jointly shape development processes (Fürst, 2005: 23; van den Berg, 2005: 76).

2.3 Governance levels of instrument implementation or application

Being legal or administrative acts, regulatory instruments are implemented, according to the constitution of the respective countries. Even though the principle of subsidiarity is popular in many countries with social market economy, **regulation of technology and performance** are tasks mostly taken up by **national level** authorities – sometimes in fulfilling supranational obligations, e.g. by the EU. In federally organised nations, such as Germany, competencies to specify and further develop regulation may be passed on to state, regional or local level.

Spatial planning, in contrast to the two other types of 'direct regulation', often (re-)focuses on more decentralised levels of governance, namely the **regional and local government levels** (Fürst, 2005: 24; Steelman, 2001: 71). According to the principle of subsidiarity in some countries (e.g. Germany) the authority on land-use planning explicitly resides with local governments. But its **different degrees of binding force** make it possible to apply **(spatial) planning at all governance levels** in parallel complementing one another. Therefore, examples of practical application range from international to local or even project-level.

For a better illustration, Table 1 provides some concrete examples for typical regulations at different governmental level chosen from the POLICYMIX case study sites.

Table 1. Examples of 'direct regulation' at different governmental level chosen from the POLICYMIX case study sites

Type of 'direct regulation'	Government level of implementation / application	Example
Natural resource management standards	International	Convention on Biological Diversity (CBD), EU Water Framework Directive, EU Cross Compliance Standards
	National	Nature Conservation Laws, Good Agricultural Practices, Proper Forest Management Rules (e.g. Forest Acts at German National State Level)
Species protection	International	Convention on International Trade in Endangered Species of wild fauna and flora (CITES), Convention on the conservation of Migratory Species of wild animals (CMS), EU Habitats Directive, EU Birds Directive
	National	Nature Conservation Acts, National Species Protection Acts, Game Laws
Protected areas	International	Ramsar-Convention on Wetlands, EU Natura 2000 network
	National	National Parks and Nature Reserve Sites in Germany
	Local / Regional	Local Protected Areas in Brazil
Spatial planning	International	Pan European Biological and Landscape Diversity Strategy (EU)
	National	National plans for habitat corridors (with transboundary perspective) and landscape specific overall concepts, e.g. in Germany
	Regional / Local	Regional and local development plans, supplemented by environmental plans (regarding biodiversity as well as ecosystem services) and sectoral plans, e.g. in Germany
	Sublocal level (or project based)	Environmental impact assessment in the EU, private forest management plans

Source: Own representation

2.4 Baseline

Very often, **'direct regulation' is the first instrument to be implemented** when facing environmental problems. This is mainly for three reasons: firstly it allows for a direct and timely response to an environmental threat. Secondly, policy makers often have vast experiences with this type of intervention and thirdly, an existing regulatory framework is a prerequisite for using incentive-based instruments such as taxes, subsidies or tradable permits. The underlying concept of 'direct regulation' is to differentiate between legal (activities or emissions covered by the standard) and illegal actions (technologies not equivalent to BAT or emissions beyond the standard). Thus, **'direct regulation' clarifies property rights** attached to a resource, e.g. land-use rights of landowners, management practices for farmers to comply with, or the requirement to offset for environmental impacts, thereby opening up the operational space for market-based instruments. Hence, it is very difficult to establish a baseline against which the conservation gains derived by implementing 'direct regulation' could be judged.

Spatial planning is usually an initial instrument too, as policies and thus specific regulatory instruments are often being considered as output of planning. It is a standard component of spatial planning documents, to declare the intention to put (possibly a set) of different, inter-coordinated measures into practice aiming at reaching the goals formulated in the plan. But planning has to consider both legal regulation as well as plans from other governmental levels (and sometimes other sectors too).

2.5 Range of application of instrument

As stated above, 'direct regulation' is still **the most widely used approach for environmental protection**. 'Direct regulation' is applied both, in terms of special conservation instruments (e.g. acts and plans for nature conservation) as well as in terms of sectoral or cross sectoral policy integration of nature conservation.

With regard to biodiversity conservation, the contribution of sectoral regulations should not be underestimated. In agriculture, for example, regulating fertiliser use can reduce nutrient run-off into soil and water, prevent eutrophication in river systems, lakes and coastal areas and algae build-up on beaches. 'Direct regulation' of this type thus very often support multiple ecosystem services and benefits, such as touristic, aesthetic and cultural values, reduced health impacts, preservation of provisioning and regulating services (e.g. carbon storage of soils). Table 2 gives an overview for selected sectoral 'direct regulation'-measures and the ecosystem services that may be affected.

Table 2. Examples of 'direct regulation' that benefit ecosystem services

Regulated activity	Type of regulation	Affected ecosystem service
Water use	Drinking water Water / groundwater extraction Waste water treatment Water body condition Water pollution and quality	Fresh water Food Water purification Water regulation Natural hazard regulation Recreation and ecotourism Aesthetic values Water cycling Nutrient cycle
Air pollution	Ambient air quality standards Emission standards Off-gas treatment Fuel efficiency standards Lead ban motorfuels Exhaust emission standards	Food Fresh water Air quality regulation Climate regulation Natural hazard regulation Recreation and ecotourism
Land use	Spatial planning / zoning Mineral extraction Soil protection and contamination	Food Fiber Fresh water Biochemicals Water regulation Climate regulation Natural hazard regulation Erosion control Air quality regulation Aesthetic values Cultural Diversity Recreation and ecotourism Soil formation Water cycling Nutrient cycle

Regulated activity	Type of regulation	Affected ecosystem service
Agriculture	Required minimum practices Best practices Fertilizers Regulation on transgenic crops	Food Fiber Climate regulation Erosion control Pest control Disease regulation Recreation and ecotourism Soil formation Nutrient cycling
Forestry	Afforestation / Reforestation Best practices Timber harvest regulation, e.g. allowable cuts Forest product licensing Hunting licensing Abstraction of non-timber-forest	Food Fiber Biochemicals Climate regulation Erosion control Natural hazard regulation Water regulation Aesthetic values Recreation and ecotourism Inspiration Water cycling Nutrient cycle
Fisheries	Catch licensing Nursery protetcion Mesh size	Food Genetic resources Climate regulation Recreation and ecotourism Nutrient cycle
Nature Protection	Protected areas Protected Species Act Habitat Directive Birds Directive	Fresh water Genetic resources Biochemicals Natural hazard regulation Aesthetic values Inspiration Educational value Spritual and religious values

Key: Provisioning Services; Regulating Services; Cultural Services; Supporting Services

Source: Hansjürgens et al. (2011b: 306 ff.)

3 Literature review of instrument performance

On a general level, **evaluating biodiversity conservation policies is challenging** for two reasons (Moran et al., 2010: 826): firstly, the understanding of the ecological significance of different species and habitats is incomplete and, secondly, conservation sciences has no strong tradition for evaluating outcomes. Nevertheless, since resource for biodiversity conservation are severely limited, understanding both the (economic) benefits and costs of conserving ecosystems will help to allocate resources efficiently (Naidoo and Ricketts, 2006: 2153). The same argument holds for the question which instrument to employ in order to conserving biodiversity efficiently or to reach environmental goals cost-effectively.

When **reviewing the impact of 'direct regulation'-instruments** it has to be acknowledged that any attempt to assess the measurable results of this type of intervention must deal with **limited and uneven information** (Fiorino, 2006). Economic research has focused on regulatory approaches to environmental preservation only occasionally. Very often command and control-regulation is simply used as backdrop of demonstrating the superiority of more flexible approaches, such as market-based instruments.

Systematic evaluation is also lacking for spatial planning (Berke et al., 2006: 581f.; Brody and Highfield, 2005: 160; Laurian et al., 2010). The main reason for this is the complexity of the planning process in combination with its preconditions (both regarding the phase of plan development as well as plan implementation), which is leading to reasonable methodological challenges (Talen, 1996a: 249). These methodological challenges can be summarised to three main problems (see Fürst, 2005: 19):

- **Attribution problems**: It is difficult to identify cause-effect-relations between planning and environmental outcomes, because planning is usually controlling only a few of many (potential) influencing factors (see Fürst, 2005: 19). One challenge in this context is to find out if a measurable change in the landscape (or that the landscape remained unchanged) is indeed an impact of planning. Furthermore and closely related, it is hard to decide on a baseline against which the impact of planning could be judged, e.g. question how the landscape would have changed without planning intervention. In this context a comparable counter fact is generally missing for plans which are set up for a certain planning unit individually, e.g. project-based development plans for infrastructure facilities.

- **Indicator problems**: There is disagreement among planning scholars over how to define and measure the success of planning. Alexander and Faludi (1989) for example reject a pure 'means-ends approach', i.e. to which degree decisions, outcomes, or impacts adhere to the objectives, instructions, or intent expressed in a plan. Indeed, policies formulated in plans have to be modified in response to uncertain political and socioeconomic conditions. Additionally, when using the objectives of the plan as a benchmark for plan outcome, one has to consider that these objectives are already integral part of the plan itself. Their definition is the second step and a result of the planning process, which means that it is to question whether the formulated goals are appropriate evaluation criteria. Furthermore, the value of planning should be measured by more than plan content alone. Exchange of knowledge among actors or improved competence in jointly solving problems (social learning) can be vital outcomes of the planning process, too. Hence, in particular quantitative measures are mostly rejected in evaluating impacts of planning (see Fürst, 2005: 19; Talen, 1996a).
- **Time frame problems**: (Spatial) planning tends to be a long-term policy instrument and it remains often unclear "when the outcome of a plan should be determined" (Brody and Highfield, 2005: 160). This is why it is often difficult to establish a time frame for evaluating the success of planning.

Despite the manifold obstacles to evaluating regulatory instruments, the following aims to provide an overview about the normative insights and the empirical evidence available to evaluate 'direct regulation'. It strives to analyse the (1) environmental effectiveness and (2) the cost-effectiveness of 'direct regulation', (3) its distributive impacts and lastly (4) the institutional requirements and prerequisites needed for a successful implementation.

3.1 Environmental effectiveness

For the purpose of this literature review, environmental effectiveness shall be understood as consisting of three major questions. Firstly, was the environmental goal reached by the use of the instrument (e.g. reduction of pollution, increase in protected area etc.)? Secondly, how long did it take till the instruments had an effect? And thirdly, did the positive effects of the instrument last?

The following strives to answers these questions by examining the available literature that evaluates 'direct regulation'. Starting from the normative observations on the effectiveness of this instrument group, the empirical evidence is analysed.

3.1.1 Normative findings on the effectiveness of 'direct regulation'

From a theoretical standpoint, 'direct regulation' is perceived as a **highly effective tool for environmental conservation**. This is mainly because of four reasons:

- Firstly, by prohibiting certain action, stopping environmentally harmful activities to safeguard a minimum standard of conservation becomes possible in rather short time.
- Secondly, the effectiveness of a standard is independent of (perceived) opportunity costs. While this may reduce the cost-effectiveness of 'direct regulation', it is an important characteristic in order to

prevent hazardous events, if perceived opportunity costs of operation do not adequately reflect all potential damages. For example, there may be a substantial uncertainty about the long-term environmental and health impacts of genetically modified organisms, oil spills or nuclear accidents. Under these conditions, incentive-based approaches, such as taxes, will fail to assure a 'safe minimum standard' or to avoid irreversible losses, e.g. in biodiversity supporting vital ecosystem services.

- Thirdly, 'direct regulation' can be adapted to consider local conditions. For example, for non-uniformly mixed pollutants it may be necessary to differentiate standards between polluting sources or activities, e.g. by zoning-rules for spatial development or by different standards being applied to sources affecting different immission areas. Moreover, this may be easier done by 'direct regulation' than by differentiated tax rates (which may not be legally allowed) or a spatially explicit designed permit market (so called ambient trading, which may cause prohibitive transaction costs, see Atkinson and Tietenberg, 1984)
- Lastly, in contrast to taxes or payments, the allocative power of 'direct regulation' is not limited by the incentive structure of the policy addressees: the steering power of taxes may decline when inflation marginalises tax rates (e.g. with taxes that are surcharged to retail prices) or if changes in technology erode the tax basis (e.g. for taxes on certain pollutants like lead).

On the other hand, there are **characteristics distinctive of 'direct regulation' that may hamper its effectiveness**. Firstly, 'direct regulation' is prescriptive and is thus leaving addressees little room to manoeuvre. Hence, there will be high political resistance to set tight standards that might cause high compliance costs. In a dynamic setting it will be (politically) costly to tighten the standard or agree on a new (and more advanced) best available technique or management practice. Secondly, it is unclear, whether the set standard or best technique is indeed the environmentally friendliest way of managing biodiversity and natural resources. As any other policy instrument, 'direct regulation' is a child of political debate and it is highly likely that other than only environmental concerns influence the final decision on the standard or plan. And lastly, as emissions below the standard or environmental impacts caused while complying with the prescribed technical norms are not burdened by 'direct regulation', there is no incentive to further reduce environmental impact (as would be the case in environmental taxation or permit trading).

3.1.2 Evidence on the effectiveness of 'direct regulation'

The question now is, whether the **empirics of 'direct regulation' can underpin these claims**. Soares-Filho et al. (in press) state that protected areas now shelter 54% of the remaining forests of the Brazilian Amazon and contain 56% of its forest carbon. They found that three major categories of **protected areas** (indigenous land, strictly protected, and sustainable use) showed an inhibitory effect in reducing carbon fluxes to the atmosphere from deforestation and their associated costs. Of 206 protected areas created after 1999, 115 showed increased effectiveness after their designation as protected. The recent expansion of protected areas in the Brazilian Amazon was responsible for 37% of the region's total reduction in deforestation between 2004 and 2006 without provoking leakage.

Wätzold (2004) shows, that the implementation of an **SO_2-emission standard** in Germany helped to immensely reducing air pollution. So called 'Waldsterben', a phenomena that perceived huge public interest in the 1980s in Germany was mainly caused by 'acid rain', which was an effect of the enormous SO_2-emissions from energy-producing combustion plants. Germany therefore set a tight SO_2-emission standard at 400 mg/m^3 that all plants had to comply with by 1993. Following the enactment of the standard, the electricity sector embarked upon a major reduction programme that led to sharp decline in SO_2-emissions from way over 2000 mg/m^3 in 1982 to only 154 mg/m^3 in 1995. A similar example is reported by Fiorino (2006: 59 ff.). The author found the emissions of many common air pollutants to have declined significantly in the US since the early 1970s, mainly because of regulatory emission standards. Phasing out chlorofluorocarbon end of the 1980s / early 1990s also marked a success story of environmental regulation.

Sometimes **hazardous events triggered successful 'direct regulation'** (Hansjürgens et al., 2011a), such as:

- the industrial accident near Seveso, Italy in 1976 that triggered the introduction of EU Seveso Directive to improve security in handling dangerous substances;
- the oil spill of tanker 'Erika' near the French Atlantic coast in 1999 that triggered the legislation for double hulled ships and the EU Liability Directive; or
- the 2000 pollution of Danube River that triggered the EU Mining Waste Directive as of 2006.

To cite an example from **forestry policy**, the decline of Swedish forests during the 1980s and 1990s led to the Swedish Forestry Act being updated in 1994. The new act provides management standards for forest lots and set quotas for maximum allowable cut per annum to promote an even age distribution of forest stands. Recent statistics prove that the regulation has had positive results as the number of old or deciduous trees recovered over the past 20 years (increase of 10 to 90%, depending on diameter) (see Hansjürgens et al., 2011b and the sources cited there).

Laycock et al. (2009) assessed the **effectiveness of the UK Biodiversity Action Plan** (UK BAP) implemented in 1994. They found that nearly half of all studied Species Action Plans (SAP) that were launched as a consequence of the UK BAP, achieved an effectiveness of 100%, while 10% of all studied SAPs have achieved zero effectiveness at meeting their targets.

Nevertheless, there are also authors who claim that **conservation regulation may be a backward step for biodiversity**. An analysis by Harrop (1999), particularly from the UK and European perspectives, found that the way in which the law deals with the conservation of species and habitats has the potential to obstruct the comprehensive preservation of biological diversity. However, this is mainly attributed to the emphasis of one species to the detriment of others and to the failure to address comprehensive inter-species and habitat relationships (Harrop, 1999: 679).

Although protected areas are unarguably beneficial in conserving biological diversity, Kharouba and Kerr (2010) show that this approach is not a silver bullet to all threats lurking. Kharouba and Kerr compare rates of butterfly species richness and composition change within protected areas against distributions of randomly selected, ecologically similar, but non-protected, areas in Canada. Change in species richness and composition within protected areas were, for the most part, the same as changes observed among random areas outside protected area boundaries. The authors thus argue, that existing protected area networks in Canada have provided **little buffer against the impacts of climate change** on butterfly species richness, possibly because land-use change surrounding long-standing protected areas has not been substantial enough to elevate the habitat protection afforded by these protected areas relative to other areas.

A mixed result is obtained by Craigie et al. (2010) in analysing the performance of 78 African **protected areas in maintaining populations of large mammals**. The index compiled by the authors reveals on average a 59% decline in population abundance between 1970 and 2005. Indices for different parts of Africa demonstrate large regional differences, with southern African protected areas typically maintaining their populations and western African areas suffering the most severe declines. These results indicate that African protected areas have generally failed to mitigate human-induced threats to African large mammal populations, but they also show some successes. Shepherd and Nijman (Shepherd and Nijman, 2008) use the trade in bear parts from Mayanmar to illustrate the ineffectiveness of the enforcement of international wildlife regulations.

In summary, **evidence available seems to support the theoretical claim that approaches of 'direct regulation' are an effective tool** for environmental protection. This is especially true, when the environmental status is perceived as very bad hitherto and immediate efforts are claimed necessary to improve environmental status.

Due to its different levels of binding force, the claim of theoretically high effectiveness cannot in general be transferred to (spatial) planning. It has to be taken into account, that planning evolved in the last centuries, where nowadays layers of older and newer philosophies of planning exist in parallel (Fürst, 2005). Another reason for seeing (spatial) planning in a nuanced light is that it is diverse, also because of the variation in administrative systems and the broad range of application of (spatial) planning across governmental levels and sectors. And finally, the effectiveness (and also efficiency) of planning depend very much on the problem to solve along with its institutional and social contexts (see also section 3.4).

Against this background, it is useful to differentiate between planning philosophies under different preconditions, when assessing environmental effectiveness of planning. The evolution of planning (and its social relevance) during the last centuries can be considered as a first indicator in these matters. The observable change in planning philosophies can be understood as a reaction on implementation problems planning faced. These implementation problems again were resulting from changes in social, political and governmental contexts, which were leading finally to changes in the concept of the state and government in general.

The first five decades of the last century, (spatial) planning was carried out according to the 'God-the-father-model'. Planners steered spatial development technocratically towards governmental objectives (Fürst, 2005). This planning philosophy was based on the strong position of the state and was carried on even to the 1970s. While people had confidence in a 'higher rationality' and expertise of governmental agencies and asked for balancing the capitalistic economic system, a 'planning-euphoria' could be registered at the end of the 1960s (Fürst, 2005: 20). Methodological professionalism and specialisation of planning increased and a 'scientification of planning' took place as believe in the regulatory power of planning was strong (Fürst, 2005: 20)[2].

In the wake of the devaluation of the US Dollar (1971) and the "oil crisis" (1973/74) this belief, that the state was able to control and manage (economic) development from a centralistic position turned (see Fürst, 2005). So, in the 1980s a fundamental shift in planning paradigms took place. Planning professionals began to deliberately understand "planning as a political process" (see Fürst, 2005). Simultaneously, a move towards participatory and community-based approaches could be observed, which were applied at more decentralised levels of decision making (Steelman, 2001: 71). Furthermore, challenged by privatisation, deregulation, speeding up of governmental procedures and the concept of the cooperative and activating, state planners started trying to activate forces of social self-regulation and public private partnerships (Fürst, 2005: 20).

Based on this development, two planning philosophies can be contrasted: the rational planning philosophy and the communicative planning philosophy, which are the two opposite poles of a continuum of recent planning approaches.

[2] *This development distanced "the common person from the language, processes, and institutions where decisions were debated and occurred. While relevant for many policy areas, this trend especially was true in the realm of environmental and natural resource policy" (Hill, 1992 in Steelman, 2001).*

Regarding these two approaches Burby (2003) found that greater involvement of stakeholders in the planning process significantly improved implementation success. Berg (2005: 76) also states, that for modern planning "mediating and moderating with modesty is more fruitful than acting from a powerful position".

Steelman (2001: 71f.) however discussed system immanent advantages and disadvantages of both approaches based on theory. Using a plan for the Monongahela National Forest in West Virginia as an example she found that "neither an elite nor participatory model of decision-making dominated in the planning process; rather, both forms of decision-making contributed to important elements in formulating this successful National Forest plan".

Nevertheless, the few existing studies on the effectiveness of (spatial) planning suggest, that plan implementation – measured by the conformance of action to a plan and plan outcomes – is weak (implementation gap) (Berke et al., 2006: 595). On the other hand, planners are increasingly focusing on facilitating plan implementation (Fürst, 2005; van den Berg, 2005). In parallel to the two planning philosophies, two styles of plan enforcement can be distinguished (Berke et al., 2006: 586):

- "a deterrence style, which emphasises a strict interpretation of plan policies, a reliance on legalistic and punitive rules, a minimal provision of technical information and assistance, and written rather than verbal modes of communication in processing permit applications (…)"; and
- "a facilitative style, which emphasises a flexible interpretation of policies, the provision of technical assistance, and verbal modes of communication."

Several studies suggest that the facilitative style of enforcement is more effective than strict coercion, regulation, and an overreliance on regulatory enforcement strategies (see e.g. Burby et al., 1998; Dalton et al., 1989; Volker, 1997). In contrast, Berke et al. (2006) showed that in cases where the addressees of a plan are seeking for reliable conditions to base their decisions on, the deterrence style of enforcement can be more successful. Nevertheless, Berke et al. (2006) found also that awareness-building programmes aiming at "a better understanding of the development problems facing communities, plans and their goals and policies", i.e. the provision of information, explanation of plan policies and associated rules, as well as the conveyance of policy advice or the coordination with other public agencies led to a better plan implementation.

Besides the different philosophies of planning and plan implementation, plan quality is another important factor for the effects of planning. Based on a summary of prior studies on plan quality, Berke et al. (2006: 585) outline "four key characteristics of good plans:

(1) a clear identification of issues important to the community;

(2) a strong fact base that incorporates and explains the use of evidence in issue identification and the development of policies;

(3) an internal consistency among issues, goals, objectives, and policies; and

(4) the monitoring of provisions to track how well objectives and goals are achieved".

Summarising this discussion and the above cited example of the Monongahela National Forest Plan one can conclude that effectiveness of (spatial) planning "hinge on two factors: (a) they [plans] must be sound technically, and (b) they must be acceptable to the multiple publics that utilise them" (Steelman, 2001: 85).

3.2 Cost-effectiveness of 'direct regulation'

The cost-effectiveness-criterion can be summarised by the question, whether the environmental goal is reached at the lowest production costs possible when using the instrument in question. Production costs comprise both abatement costs of the policy addressees, in particular opportunity costs of foregone economic activities due to the regulation, and transaction costs associated with the instrument (Birner and Wittmer, 2004). Transaction costs can be borne either by the regulator (e.g. political bargaining costs, costs of implementation and finally monitoring and enforcement of addressees) or by the policy addressee (e.g. time and resources spent on understanding and adapting to the new standard or management practice). Furthermore, cost-effectiveness can be looked at in a static perspective, i.e. are production costs minimised now, as well as in a dynamic perspective, i.e. will potential production costs gains be realised by the instrument in the future?

The following describe the normative observation in the cost-effectiveness of direct regulation in both static and dynamic view, followed by a survey of the available literature evaluating practical 'direct regulation'-measures.

3.2.1 Normative findings on the cost-effectiveness of 'direct regulation'

In text book economics, the cost-effectiveness of 'direct regulation' is referred to with scepticism. Typically, 'direct regulation' is judged as 'traditional', 'inefficient' policy not able to establish least cost-solutions in (environmental) management. This can be theoretically substantiated from both a static as well as a dynamic perspective.

In a static setting, 'direct regulation' underperforms compared to incentive-based approaches, such as taxes or permit trading, because 'direct regulation' ignores differences in opportunity costs among policy addressees. All actors have to comply with a set standard, e.g. for reducing pollution or providing environmental conservation. But the environmental goal may be reached with lower compliance costs, if actors with relatively low opportunity costs would avoid more emissions or provide more environmental conservation compared to actors with relatively high opportunity costs. Secondly as stated above, environmental damages caused while complying with the standard are not burdened by 'direct regulation' but have to be accepted as there are legally allowed. Thus there is no incentive for policy addressees to reduce negative environmental impact below the standard. In contrast, taxes may be levied on / permits must be obtained to cover all emissions and may therefore be able to achieve greater emission reductions.

This characteristic of 'direct regulation' is also the main cause for their relatively weak efficiency in a dynamic setting. As 'direct regulation' provides no incentive to reduce negative environmental impacts below the standards stimuli for private policy addressees to develop more environmentally friendly techniques and methods of production are lacking. Hence, more public spending for research and development has to be provided in order to create the engineering progress necessary for further environmental improvement. Moreover, standards have to be continuously updated to correspond to the 'best available technology' or to the best management practices. Notably in regulating emissions, the more recent and often more strict standards are only applied to new facilities, whereas older sources keep falling under the (weaker) standards valid at the time the operating permission was granted. Such exceptions in implementing stricter standards further decrease the efficiency of 'direct regulation'.

The objective of strategic planning is to optimise aims and means (van den Berg, 2005: 76), which is why – from a theoretical point of view – the chosen measures will be the most efficient ones that were available and sufficient to reach the given objectives. Beyond the expenses for implementing planned measures, the main costs of planning are transaction costs. The latter ones are sometimes used as an argument against public participation (Steelman, 2001: 72), which is confronting the tendency of higher effectiveness of community-based approaches. Indeed, promoting direct participation as a strategy for a sustainable landscape development is very costly, but Buchecker et al. (2003: 44) claim that it "is worthwhile, however, if we consider the far reaching effect of this strategy". Yet, several attempts to systematically evaluate the cost-effectiveness of spatial planning systematically by means of cost-benefit-analyses, e.g. in Germany failed because these approaches were not able to cope with the complexity of planning (Fürst, 2005: 19). So, empirical evidence on the efficiency of spatial planning (and its different philosophies) is missing.

3.2.2 Empirical evidence on the cost-effectiveness of 'direct regulation'

As stated in the beginning of section 2, the **empirical basis on the efficiency or cost-effectiveness of 'direct regulation' is rather weak**. In the early discussion on how to cost-effectively deal with air pollution, 'direct regulation' was employed as a baseline for demonstrating the superiority of market-based approaches, especially emission trading (see inter alia Atkinson and Tietenberg, 1982, 1984). Only occasionally, analysis was carried out mainly on the efficiency of 'direct regulation' and emission standards.

The literature considered for this review has mostly focused on protected areas. Many authors have demonstrated that **protected areas are a worthwhile investment** from an economic perspective. The most widely cited estimate puts the benefits of an expanded protected area network (covering 15% of the land and 30% of the sea) at an annual net value of US$ 4.4 trillion compared to cost oft US$ 45 billion per year, including management, compensation for direct costs, and payments of opportunity costs for acquiring new land (Balmford et al., 2002). The TEEB study synthesised findings from seven studies that compare the benefits delivered by intact ecosystems with benefits from conversion to agriculture, aquaculture or other primary production. Including major market and non-market values, the global benefits from protection appear to be on average of these seven studies 250% greater than benefits from conversion (see Kettunen et al., 2011 and the sources cited there). In conducting a spatial evaluation of the costs and benefits of conservation in the Atlantic forest in Paraguay, Naidoo and Ricketts (2006) found that benefits exceeded costs in areas with carbon storage dominating the ecosystem service value and swamping opportunity costs.

Another strand of literature has dealt with the **cost-effective selection of reserve sites** and the advantage of integrating economic costs into conservation planning. Studies that incorporate the spatial distribution of biological benefits and economic costs in conservation planning have shown that limited budgets can achieve substantially larger biological gains than when planning ignores costs (Naidoo et al., 2006). In another study, Naidoo and Iwamura (2007) found that conservation plans that consider costs represent endemic species at 10-33% of the opportunity costs of plans that do not. Moore et al. (2004) found that factoring the costs of conservation management into the planning process results in a marked increase in the cost-effectiveness of site prioritisation schemes in Africa.

Besides looking at costs, there are many studies emphasising the **benefits attached to protected areas**. Exemplarily, Schuyt (2005) and Njaya (2009) show that the protected wetlands of Lake Chilwa (Malawi) has an annual fish catch worth US$ 18 million and produces more than 20% of all fish caught in Malawi. Leuser National Park in Indonesia was estimated to generate US$ 9.5 billion total economic value between 2000-2030 from a range of ecosystem services considered (van Beukering et al., 2003). In Switzerland, large proportions of forests are managed to control avalanches, landslides and rockfalls, thereby providing services estimated at US$ 2-3.5 billion annually (International Strategy for Disaster Reduction, 2004).

Most often, economic research on biodiversity conservation has focused either on the costs of conservation reserves or the benefits of intact ecosystems; however, there are only very few studies **simultaneously considering costs and benefits of conservation**. Naidoo and Adamowicz (2005) is such an exemption: their quantification of costs and benefits of avian biodiversity at a rainforest reserve in Uganda show that the economic benefits of biodiversity exceed the costs of conservation. Moreover, they demonstrate that entrance fees if optimised to capture tourist's willingness to pay for forest visits and increased numbers of bird species could cover the budget necessary to protect roughly 90% of bird species (Naidoo and Adamowicz, 2005: 16715). Very recently, Armsworth et al. (2011) examined variation in management costs across 78 small protected areas in the UK and found 'site area' to be the most important determinant; thus demonstrating that there are 'economies of scale' within managing protected areas.

In recent years, the **implementation of the Natura 2000 network** has triggered some research on the cost-effectiveness of this type of 'direct regulation' and the covered protected areas. In Scotland, the ecosystems protected by Natura 2000 sites provide benefits to the Scottish public worth more than three times than associated costs, including direct management and opportunity costs (Jacobs, 2004). Gantioler et al. (2010) estimate the annual costs of implementing the Natura 2000 network is as high as € 5.8 billion per year for the EU-27, which is around four times higher than the annual contribution of the present EU budget. Although there is no single estimate for the benefits of Natura 2000 provided by Gantioler et al., a number of examples demonstrate that the benefits can be larger than the associated costs. In Ireland, the total rate of return on government support to the Burren Park was estimated (conservative) to be around 353 – 383%, (without or with tourism), and 235% if all operating costs of the farming programme and all direct payments are considered (Rensburg et al., 2009 cited in Gantioler et al., 2010). The exploratory study by Wätzold et al. (2010b) derives recommendations for improving the cost-effectiveness of implementing and managing Natura 2000-sites. They place special emphasis on the importance of transaction costs associated with different management types. However, at this stage they do not provide any specific judgement on the cost-effectiveness of the regulatory approach as such.

The study done by Eppink and Wätzold (2009) focussed on the **costs associated with species protection** induced by the EU Habitats Directive. In response to that obligation, Member States have implemented a variety of conservation measures including the rejection, modification or delay of land development plans. Although the paper has its focus on methodological issues the case study on protecting habitats of the European Hamster in Germany revealed that in particular the costs of modifying development plans can be substantial. Their estimate put these costs to be between € 19.6 and € 38.2 million, whereas discounted costs of compensation payments to farmers (€ 214,453 to € 263,647) and costs of conservation management (€ 769,101 to € 924,881) represent only a small fraction of total conservation costs (Eppink and Wätzold, 2009: 807).

In summary, empirical evidence available yet is not sufficient to prove the theoretically derived claim of weak efficiency / cost-effectiveness of regulatory approaches in the field of biodiversity conservation, though it might be possible to disprove the claim in particular contexts. This might at least partly be explained in missing research interest in regulatory approaches in general. However, and more important, as 'direct regulation' typically is the pioneering approach to control environmental problems and is often pursuing multiple goals, it elude itself from cost-benefit analysis based on a clearly defined 'means-ends approach'. Nevertheless further evaluation is needed, building on a framework that is able to deal with the multifaceted benefits (and costs) associated with this type of intervention.

3.3 Social and distributional impacts of 'direct regulation'

By the implementation of 'direct regulation', **property rights –** negligible whether prior existent or not – are (created and) formalised. This may impact the allocation of ecosystem services, e.g. access to water, wood collected in the forest or the way in which resources are used, e.g. obligations to apply a certain technique or to offset any negative impacts on ecosystems. A serious drawback in this respect, notably in the case of creating protected areas, is the potential of precluding traditional but not officially acknowledged property rights, e.g. by indigenous people.

Legal instruments are typically following a **top-down-approach** in design and implementation. Hence, influence of stakeholders via participatory efforts is curtailed to lobbying in political decision-making. Spatial planning, in contrast, involves usually both, elements of top-down approaches as well as elements of **bottom-up approaches** (principle of countervailing influence). Moreover, the institutionalised process of deliberation and balancing by means of planning can be seen as a precondition for including stakeholders when designing policies (see Buchecker et al., 2003).

As standards (at least in theory) do not distinguish between the actors addressed, they force an **equal share in reducing environmental impacts**, thereby fulfilling the 'polluter pays' principle. However, as environmental impacts below the standard are not penalised, this is true only partially.

Although not all benefits are (or can be) monetised, there is ample **empirical evidence**, notably for protected areas, that local stakeholders might benefit from the ecosystem services maintained within the reserve sites, depending on the distribution of derived profits. Andam et al. (2010) estimate **impacts of protected area systems on poverty** in Costa Rica and Thailand and find that although communities near protected areas are indeed substantially poorer than national averages, an analysis based on comparison with appropriate controls does not support the hypothesis that these differences can be attributed to protected areas. In contrast, the results indicate that the net impact of **ecosystem protection was to alleviate poverty**. The study conducted by Sims (2010) demonstrate that in Thailand protected areas increased average consumption and lowered poverty rates, despite imposing binding constraints on agricultural land availability. The gains are likely explained by **increased tourism** in and around protected areas. However, net impacts are largest at intermediate distances from major cities, highlighting that the spatial patterns of both costs and benefits are important for efforts to minimise conservation-development tradeoffs. In Indonesia, people living near intact forests protected by Ruteng Park have **fewer illnesses** from malaria and dysentery, children miss less school due to sickness and there is less hunger associated with crop failure (Pattanayak and Wendland, 2007). In Cambodia's Ream National Park, estimated benefits from sustainable resource use, recreation and resource are worth 20% more than benefits from current destructive use. The distribution of costs and benefits favours local villagers, who would earn three times more under a scenario of effective protection than under a scenario without management (De Lopez, 2003). Lastly, in Caprivi Game Park, Namibia, sustainable harvesting techniques of palms enabled local women to supplement household incomes by selling woven palm baskets to tourists. Producers grew from 70 in the 1980s to more than 650 end of 2001 (World Resource Institut, 2005).

Mayer et al. (2010), focusing on the revenues of ecotourism in six national parks in Germany, found that **protected areas can create considerable income for adjacent communities**. The authors found that the total impact of tourism ranges between € 525 million and € 1.9 million per annum, reflecting the national parks' distinct trajectories as tourist destinations. Point Pelee National Park in Canada attracts over 200,000 visitors and birdwatchers annually, who bring millions of dollars of additional revenue into the local area (Hvenegaard et al., 1989).

There is also evidence, that more **cost-effectively selecting reserve sites may increase social distribution** of costs and benefits of protected areas. A recent study by Adams et al. (2010) concluded, that planning with opportunity costs to single stakeholder groups can result in cost burdens to other groups that could undermine the long-term success of conservation. Thus, they argue for the necessity of an understanding of the spatial distributions of opportunity costs that are disaggregated to groups of stakeholders in order to make informed decisions about priority conservation areas.

In summary, though a comprehensive analysis of the social impacts of 'direct regulation' is missing, there is ample evidence, notably on protected areas, that regulating use of natural resources and protecting reserve sites to maintain ecosystem service provision have positive social impacts, especially on the rural poor.

3.4 Institutional context and requirements

In order to be effective, 'direct regulation' needs **monitoring, enforcement and powerful sanction mechanism** in case of non-compliance. However, this is not a sole characteristic of 'direct regulation' but a prerequisite for each and every intervention to become effective. By contrast, 'direct regulation' might be easier to implement than innovative market-based instruments because **policymakers have long-lasting experience** with this type of intervention; also policy addressees are used to this kind of instruments and might more easily adapt to.

Again, most literature is available in the context of protected areas. Dustin Becker (2003) found that substantial political autonomy, stable economic conditions, land tenure security and a culture of trust and collective concern are usually critical for the success of indigenous or community-based conservation. Barrett et al. (2001) argue that successful institutions in charge of managing protected areas need authority, ability and willingness to promote sustainable use of resources, facilitate equitable distribution of costs and benefits and support different governance types. Moreover, successful establishment and management of protected areas require mechanisms for coordination and collaboration between different institutional levels, e.g. different sectors, stakeholders and government agencies (Kettunen et al., 2011).

The success of 'direct regulation' is often **a matter of resources**. The German Advisory Council on the Environment (2007) devoted a whole special report on the importance and necessity of adequate funding of monitoring and enforcement for the success of environmental policy, and nature conservation policies in particular. Furthermore, Berke et al. (2006) proved staff and financial or hierarchical power to be important factors of success and effectiveness of spatial planning as they restrict:

- the ability of the planning agencies to implement their plans directly on their own as well as
- their capacity to foster plan implementation by making 'good plans' and pursuing supporting addressees during the implementation process.

Furthermore, conclusions from history and planning theory are that **political encouragement** and the general acceptance of governmental steering (connected to the society`s level of education) were elements of success especially for the rational style of planning and plan implementation. In reverse, this means if state interventions get weaker, spatial planning loses backing (Fürst, 2005: 16). And also Steelman (2001: 73) states that "rising education levels, increased information about the environmental impacts of resource use, and increasingly savvy protest techniques have combined to make the general public a more formidable opponent to be excluded from the decision-making processes that affect their lives." But changes in social contexts (e.g. individualisation), which can be obstacles to the rational style of planning, are not automatically fostering participation of stakeholders and the general public as the contrasting planning philosophy either. Anyhow, participatory approaches have their institutional requirements and own challenges, too. Even given the opportunity, people do not participate in planning processes automatically. They "often [do] not participate when there is nothing for [them] to react against", they "have many other competing outlets for their time (…)" and they "often fail to see that it is in their long-term interest to become involved early in a planning process, even when it involves important cultural issues like hunting rights, timber cut, and recreation choices." (Steelman, 2001: 85). This is why **public participation has to be actively promoted**. Buchecker et al. (2003) recommend creating a 'sheltered framework' for direct participation within local communities. Furthermore, promoting and successfully conducting participatory approaches requires adequate skills and resources (Buchecker et al., 2003: 44). Another precondition is that leeway is given and responsibility passed to the more decentralised levels of decision-making (Buchecker et al., 2003; Steelman, 2001; Volker, 1997). And finally – likely to be valid for all styles of planning – the **attitudes of the addressees regarding planning agencies and topics** can be major obstacles for plan implementation, possibly even exceeding generous economic incentives (see Lütz and Bastian, 2002).

Predictability and stability of the systems attached to problems tackled by planning are two further preconditions of the success of planning, because planning is a long-term instrument (see Fürst, 2008b). The more predictable and stable these systems (and the institutional settings of the planning process) are the more likely is it that successful plans can be set up. And on the other hand, the less predictable and stable the settings are, the more flexible plans have to be formulated and planning has to react (Fürst, 2005; van den Berg, 2005). These findings are supported by Talen (1996b): the external factors affecting or influencing plan implementation include the complexities of local political contexts, the degree of local societal consensus about planning issues, the degree of uncertainty and available knowledge about the issues at hand and the support (or lack thereof) for planning in terms of funding or political support.

4 The role of 'direct regulation' in a policy mix

'Direct regulation' is, as was shown above, is the **most important ingredient** of any policy mix striving for biodiversity conservation and sustainable management of ecosystem services – they form the backbone of the policy response to these challenges (Hansjürgens et al., 2011b). In particular in the field of hazard prevention (e.g. banning toxic substances that may be released to the environment) and prevention of species extinction (protection of red listed species) strong and 'direct regulation' is called for. There is a considerable scope for further usage of 'direct regulation' - measures; however, a strong regulatory framework, for example well defined and enforceable property rights, also constitutes a **basic precondition for introducing other instruments**, such as offset requirements, biodiversity banking, and payments for ecosystem services or ecological taxes.

'Direct regulation' can be highly effective in biodiversity conservation; however, because such measures are largely ignoring opportunity costs of affected policy addressees, their efficiency or **cost-effectiveness can be augmented** by the combination with more flexible, incentive-based approaches that aim to increase acceptability of and compliance with 'direct regulation'. On a conceptual level, there are many examples available that show the potential of such a policy mix e.g. for controlling air pollution (Lehmann, 2010; Sorrell and Sijm, 2003; Tietenberg, 2003) or land development (Henger and Bizer, 2010; Korthals Altes, 2009; Nuissl and Schröter-Schlaack, 2009).

Policy mixes offer opportunities to **address various ecosystem services and various actors** in a number of different locations at the same time. The optimal mix will depend on the state of the resource in question and its characteristics, as well as the number and variety of actors affected (Hansjürgens et al., 2011b). Ecosystems close to a threshold of irreversible change might require 'direct regulation', such as bans and strict standards, whereas for the sustainable management of renewable resource, such as timber harvest or fisheries management, market-based approaches merit serious consideration. However, even for a single resource, a policy mix is often suitable. E.g. in fisheries policies, no-take zones such as marine protected areas might be appropriate to provide undisturbed spawning grounds while fish catch might be managed through tradable quotes most cost-effectively.

There is also some **empirical evidence** available, on how biodiversity conservation might be more effectively or more efficient achieved by employing several instruments in a policy mix. In the Kakamega Forest, Western Kenya, protected areas were established in order to preserve the forest's unique biodiversity from being converted into agricultural land. Nonetheless, recent research shows that degradation continues at alarming rates. Börner et al. (2009) demonstrate that an incentive scheme might balance local demand for subsistence non-timber forest products against conservation interests. Their findings suggest that a more flexible approach to determining the price of recently established forest product extraction permits would greatly enhance management efficiency without significantly compromising local wellbeing.

The creation and maintenance of nature reserves in agricultural landscape is often hampered by both ecological and economic factors, while the ecological effectiveness of agri-environment schemes (AES) is still being queried. The study by Leng et al. (2010) examined how the spatial pattern of nature reserves and AES affects the diversity of 25 target species of conservation interest in ditch banks. They studied target species plant diversity on 92 ditch banks under AES and on 102 banks not under such a regime; all of them running parallel to nature reserves. On non-AES ditch banks there was a significant decline in species richness with increasing distance from the nature reserve while this was not the case for AES ditch banks, thereby their results indicate that synergy between nature reserves and AES can enhance plant diversity.

Spatial planning can already be understood as a policy mix in itself, as it combines instruments with different binding force and is applied across governmental levels and sectors complementing one another. Berke et al. (2006: 596) conclude, that plans should "include shared governance arrangements that help planners, permit applicants, and the public to understand and account for the broad connections with federal, state, and regional programmes and policies that affect land use and development".

Within a policy mix the role of planning can be varying. Recent trends in contextual settings press spatial planning to integrate market-based instruments (pricing of scarce natural resources, property rights trading, emission trading, etc.), to collaborate more intense with the addressees, and to closer link spatial planning to spatial development schemes (strategic planning) to be successful (Fürst, 2005: 16; van den Berg, 2005: 76). These approaches of "strategic planning and interactive working have proved to be more effective in implementation" (van den Berg, 2005: 76). Further details about examples on how economic instruments as well as monitoring and sanctioning foster the performance of planning significantly can be found in Brody and Highfield (2005: 172).

In line with this, Lütz and Bastian (2002) showed, based on an analysis of local landscape plans in Saxony (Germany), that a closer link between agricultural subsidies and local landscape plans hold the potential to improve the performance of both. The EU tends to allocate agricultural subsidies more target-oriented by tying them to plans regarding for example the Natura 2000 network, watershed management plans and so on (Council of Europe, 2005).

Other example for possible roles of planning in a policy mix can be found in the agricultural and forestry sector, where (spatial) planning has been introduced into private land-use management. This is to strengthen rationality as well as long-term perspective and awareness about community values in the decisions of land users and therewith sustainability of land management. In the agricultural sector in several EU countries, farmers can receive payments for establishing environmental whole farm management plans or environmental management systems (BMLFUW (Bundesministerium für Land- und Forstwirtschaft; Umwelt und Wasserwirtschaft), 2010; Welsh Assembly Government, 2010). Furthermore, some European countries (see again Welsh Assembly Government, 2010) allocate agri-environmental payments based on whole farm management plans. By connecting these two measures public authorities intend to allocate subsidies more targeted and to increase the cost-effectiveness. First evaluations of the application of environmental whole farm management plans (EF plans) in New Zealand reveal a widespread use and development of this kind of plans throughout the country, attesting “that many regional authorities consider EF plans to be effective tools for achieving their sustainable resource management objectives” (Manderson et al., 2007: 330).

5 Concluding remarks

The term ‘direct regulation’ covers a broad range of measures, thus making it a challenge to derive some general conclusions on their performance. Despite the lack of instrument evaluation in this field, this survey has revealed some tendencies. In text book economics, ‘direct regulation’ is perceived as a highly effective tool for responding to an environmental threat; and the available empirical evidence is able to underpin this claim. In contrast, the efficiency or cost-effectiveness of the regulatory command and control-approach is scrutinised frequently in theory. Again, the analyses available on how to design regulatory conservation measures more cost-effectively, e.g. selecting reserve sites or managing protected areas, point to the fact, that efficiency of this type of intervention can be increased. The evidence on social and distributional impact of ‘direct regulation’ is somehow mixed. On the one hand, ‘direct regulation’ clarifies property rights, thereby making use or access rights legally enforceable. On the other hand, there is a risk of precluding well established, but informal property rights, e.g. those of indigenous people. The ‘polluter pay’-principle is not implemented to its full extent, as the environmental impact below the standard is not penalised but free of charge to the polluter. Finally, policy makers have far reaching experience with this type of intervention, thus making it possibly easier to decide upon and implement in practice. Its effectiveness, however, depends on well-functioning institutions for monitoring, enforcement and sanction in case of non-compliance. By its potential to effectively safeguard a minimum standard of conservation, ‘direct regulation’ it is the most important ingredient for a policy mix to conserve biodiversity and maintain flows of ecosystem services. However, as ‘direct regulation’ largely ignores differences in compliance costs among policy addressees, there is a distinct operating space for flexible, incentive-based approaches, such as taxes, permit trading or self-regulation.

The effectiveness of (spatial) planning is very much context dependent. Recent trends in (spatial) planning reveals that it is increasingly applied within policy mixes (or used to design them). The reliance on hierarchical approaches of planning and plan implementation have been replaced by concepts that understand planning as an ‘intermediate authority’, acting as a kind of interface between and combining economic, political and social steering systems (Fürst, 2005: 23).

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Offsets, Habitat Banking and Tradable Permits for Biodiversity Conservation

Rui Santos, Pedro Clemente, Paula Antunes, Christoph Schröter-Schlaack and Irene Ring

Summary

Trading permits and offsets schemes are seen as promising instruments for biodiversity conservation and their application is increasing all over the world. A review of several case studies where tradable permits, offsets or habitat banking schemes were implemented was performed analysing their main characteristics, in terms of ecological effectiveness, economic efficiency, social impacts, actors involved and the institutional context of application. The role of these instruments in a policy mix aiming at biodiversity conservation, with a special focus on potential gains resulting of complementarities with other instruments and the problems of overlapping with the ones already in place is also discussed.

1 Introduction and definitions

This chapter presents a literature review on offsets, habitat banking and tradable permits as instruments for biodiversity conservation. It is developed in the scope of work package 2 “Review of policy instruments and their roles in a policy mix” of POLICYMIX FP7-EU project. The main objective is to provide a concise synthesis of the main concepts and definitions, as well as an analysis of the key features and reported performance in relation to several selected evaluation criteria (effectiveness, cost-effectiveness, social and distributional aspects, as well institutional requirements) based on a sample of case studies. It intends to be a contribution to find some generic characteristics of this instrument, namely as a basis for a judgment about the complementarity with other instruments within a policy mix for biodiversity conservation.

Besides the available literature on the issue, twelve case studies were reviewed, including five cases from the Business and Biodiversity Offsets Programme (BBOP), namely in Madagascar (The Ambatovy Project), Ghana (Akyem Gold Mining Project), USA (Bainbridge Island), South Africa (Potgietersrust Platinums Limited (PPRust)) and in New Zealand (Strongman Mine). The remaining cases include the US Wetland Banking, the Conservation Banking (USA), Biobanking, Bushbroker (Australia), the CDC Biodiversité (France), the National Grassland Biodiversity Programme – NGBP (South Africa) and the Voluntary Malua Biobank (Malaysia).

Biodiversity offsets are defined by the Business and Biodiversity Offsets Programme (2009) as: “measurable conservation outcomes resulting from actions designed to compensate for significant residual adverse biodiversity impacts arising from project development and persisting after appropriate prevention and mitigation measures have been implemented. The goal of biodiversity offsets is to achieve no net loss, or preferably a net gain, of biodiversity on the ground with respect to species composition, habitat structure and ecosystem services, including livelihood aspects”. So, they intend to compensate for residual environmental impacts of planned developments, but after appropriate actions have been taken to avoid, minimise or restore impacts on site. It is important to share a strong agreement, among all stakeholders, about what can and cannot be offset.

Crowe and ten Kate (2010) identify two different approaches of biodiversity offsets:

- Voluntary biodiversity offsets: a developer undertakes the responsibility to compensate in circumstances where there is no legal requirement to do so, with the objective to get a business advantage (e.g. license to operate, reputational benefits, competitive advantage, market share); or
- Regulatory biodiversity offsets which are required by law.

Governments can introduce biodiversity offsetting policy and regulation in two basic ways. The first is through specific provisions on biodiversity offsets (e.g. situations of temporary loss of biodiversity such as sustainable timber harvesting) and the second is to incorporate offsetting provisions into other laws and policies, such as the environmental impact assessment (EIA), land-use planning, strategic environmental assessment (SEA), sectoral policies or broader environmental policies.

Biodiversity offsets are part of the legal framework in several countries, such as the United States of America, Brazil, Europe and Canada (ten Kate et al., 2004), and several voluntary projects are also being implemented (e.g. Australia, Uganda, Business and Biodiversity Offset Programme pilots/case studies in Ghana, Mexico, Qatar, South Africa) (Burgin, 2008; Business and Biodiversity Offsets Programme, 2009). Biodiversity offsets were used by some of the 193 government parties to the Convention on Biological Diversity as a tool to comply with their commitment of significantly reduce biodiversity loss rate by 2010 (Crowe and ten Kate, 2010). Biodiversity offsets are essentially local and bioregional tools, usually planned within the same bioregion as the area impacted, uniquely tailored to local circumstances.

The USA wetland mitigation scheme in the 1970s is considered the trigger for the concept of biodiversity offsets (Burgin, 2008; Business and Biodiversity Offsets Programme, 2009; eftec et al., 2010). Since then, it has turned global (ten Kate et al., 2004) and raised the attention of several stakeholders, such as environmental lobbies, industry (including mining, construction, oil and gas, forestry), governments and investors (International Council on Mining & Metals, 2005).

Habitat banking can, in some way, be seen as an extension of biodiversity offsets, turning offsets into assets that can be traded, creating a market system for developers. Habitat banking, also known as biodiversity trading, biodiversity banking or conservation banking is defined by eftec et al. (2010) as “a market where credits from actions with beneficial biodiversity outcomes can be purchased to offset the debit from environmental damage. Credits can be produced in advance of, and without ex-ante links to, the debits they compensate for, and stored over time”.

Habitat banking is one of the existing options for developers to offset their impacts. A main characteristic of this instrument, that distinguishes it from other offsetting instruments, is the transferability of property rights from the suppliers of the credits to the buyers. Developers, whose activity results in degradation or destruction of a natural habitat need permits that can be obtained through the restoration of a habitat with equivalent value, or by purchasing them on the market (Drechsler and Hartig, 2011). This is an advantage for developers that will stop negotiate site-by-site responses, leading to the replacement of low-performance on-site mitigation measures by meaningful financial contributions to regional conservation targets (Gillespie and Hill, 2007).

Unlike carbon, where there is a single, global metric and unit (i.e. tonnes of carbon dioxide equivalent), biodiversity credits cannot be traded internationally. However, some countries establish habitat banking schemes and designate a set of biodiversity credits as a means of defining offset requirements (Crowe and ten Kate, 2010).

Biodiversity offsets and banking go beyond traditional mitigation, like tradable permit schemes in general, such schemes encourage business to take responsibility for its impacts and help to integrate conservation objectives into mainstream business, thus helping to overcome business resistance to integrate environment and promoting a more holistic approach to business project cycles. According to Crowe and ten Kate (2010) "biodiversity offsets offer not only a risk management tool and potential business opportunity for companies, but a possible source of new and additional source of funding for biodiversity conservation and sustainable use activities".

The idea behind tradable permit schemes, in general, is to limit the use of a resource up to a politically set cap, e.g. tonnes of CO_2 emissions or hectares of agricultural land to be developed. Every unit, e.g. tonne or ha, under this cap is turned into a commodity by issuing permits allowing the holder to access or use the resource (Pruetz and Standridge, 2009). By making these permits tradable, the resources are directed towards their most profitable/efficient use, since the related projects will be able to place the highest bid for the requisite permits. Such systems can achieve two main goals of environmental policy. Firstly, resource use can be reduced to the set cap, and secondly, the costs (including opportunity costs) of reaching this target are minimised (Tietenberg, 1985).

The first large-scale attempts to use tradable permits in practical environmental policy were made as part of the US clean air policy in the 1980s and 1990s (Hahn and Hester, 1989; Tietenberg, 1990). The first attempts to use tradable permits to control land development and safeguard natural landscape date back to the 1980s and more than 140 programmes have since been launched in the US alone (Pruetz, 2003). These tradable development rights (TDRs) are applied to contain urban land development (Janssen-Jansen et al., 2009) or to protect prime agricultural land from development (Lynch, 2005). Moreover, further environmental goals addressed under a TDR scheme include providing enough land for aquifer recharge, maintaining and sustaining wildlife habitat, and minimising land fragmentation (Machemer et al., 1999). Solutions of this kind are gaining popularity across the globe and have reached e.g. China (Han, 2010; Wang et al., 2009).

2 Governance levels of instrument implementation and application

The scale and scope of biodiversity offsets is difficult to ascertain. Several forms of offset have been implemented at different levels in an increasing number of countries and sectors (ten Kate et al., 2004). The majority of published case studies are at a regional or local level (e.g. BBOP), but, for example, the US Wetland Banking (Box 1) is applied at a national scale. This means that the role of relevant governance levels varies according the conservation case under study.

In general, habitat banking and offsets require a strong involvement of the national and regional levels of governance (Bovarnick et al., 2010). At these levels, the main role is allocated to governmental departments and agencies, namely for the design and enforcement of biodiversity conservation policies and regulations which create the background to put the market working (Organisation for Economic Co-operation and Development (OECD), 2004), but their role is also relevant for monitoring and reporting (Wissel and Wätzold, 2010). Local levels of governance, including the participation of municipalities or other local authorities, are essential for this type of instruments, especially on land-use planning related issues. When the instrument is locally implemented, local governments are in addition responsible for the enforcement and monitoring of the instrument (eftec et al., 2010). Besides the active role of local governments, agencies and non-governmental organisations, local communities and landowners are also key actors. They are the suppliers of credits and the ones that assure land management and the implementation of the conservation measures.

A key aspect in habitat banking is that it contributes to reveal hidden information to the regulator (Ring et al., 2010a), by recognising that landowners have information, individually and collectively, that can be used to more effectively deliver desired environmental and natural resource management outcomes. By creating an opportunity to trade, landholders reveal important information that allows for more cost-effective use of conservation funds.

Typically, permit trading, and notably tradable development rights for landscape protection, are applied at regional level (e.g. see Kraemer et al., 2003 for an overview of water-related trading; Pruetz, 2003 for an overview of US-TDR-systems). However, permit trading in other environmental fields, namely air pollution control has also been implemented at international level (e.g. the European Carbon Trading Scheme to reduce carbon emissions, see European Union, 2003; Gagelmann and Hansjürgens, 2002), and at national level (e.g. the Acid Rain Program in the US to reduce SO_2 -emissions, see Burtraw and Palmer, 2004; Ellerman et al., 2000; Hansjürgens, 1998). In order to stimulate actual trading of permits, markets should be large enough to cover actors with sufficiently different opportunity costs (Tietenberg, 1990).

Box 1. US Wetland Banking

Wetland mitigation banks have been providing offsets in the US since the early 1980s. They became extensively used when the US Army Corps of Engineers (ACOE), the US Environmental Protection Agency (EPA), and other federal agencies published uniform guidance in 1995 on this issue (Environmental Protection Agency (EPA), 2009).

The main goal of the mitigation banks is to overcome deficiencies in the traditional mitigation approaches (e.g. permittee-responsible mitigation and in-lieu fee mitigation), and to create economic incentives for landowners to conserve wetlands or other aquatic resources (Environmental Protection Agency (EPA), 2009). This is accomplished through the transaction of credits for creating, restoring, enhancing, or protecting wetland function and value on lands protected and managed in perpetuity. A wetland mitigation bank credit is defined by EPA as a "*unit of measure representing the accrual or attainment of aquatic functions*" at a bank.

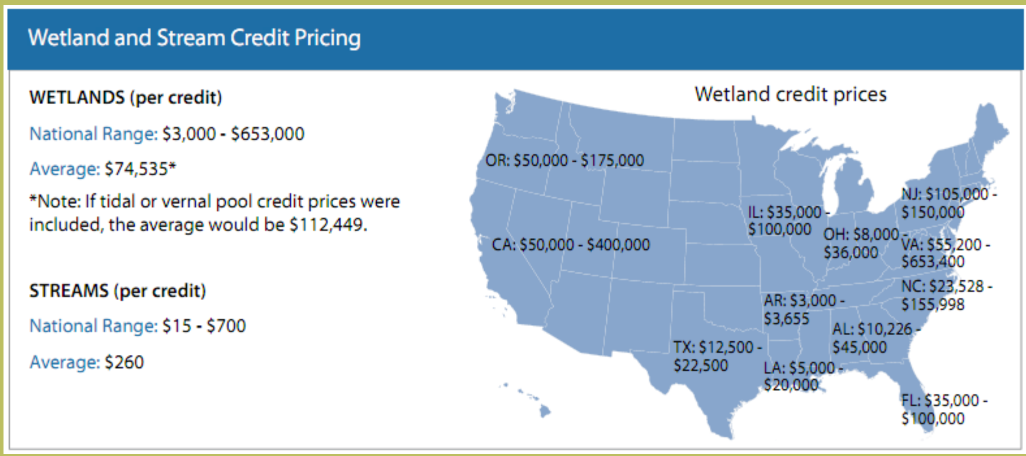
The credits are calculated differently depending on the region of the US were the bank is created. Different metrics are used, such as acres, fractions of an acre, or ecosystem functions. Therefore, it is not possible to establish any comparison between the prices of the credits from one region to another (Madsen et al., 2010). Using data from the State of Biodiversity Markets Report (2010), it is possible to display some numbers that describe the current implementation of this scheme, and also highlight the scale of the program and its regional variations (e.g. national range of credit price on figure 1).

Table 2. Mitigation Area 2008

Area of Wetland and Stream Mitigation per Annum (2008)	
Total area of wetland loss:	18,800 acres
Total area of compensatory wetland mitigation:	24,178 acres
Total linear distance of stream mitigation:	312 miles

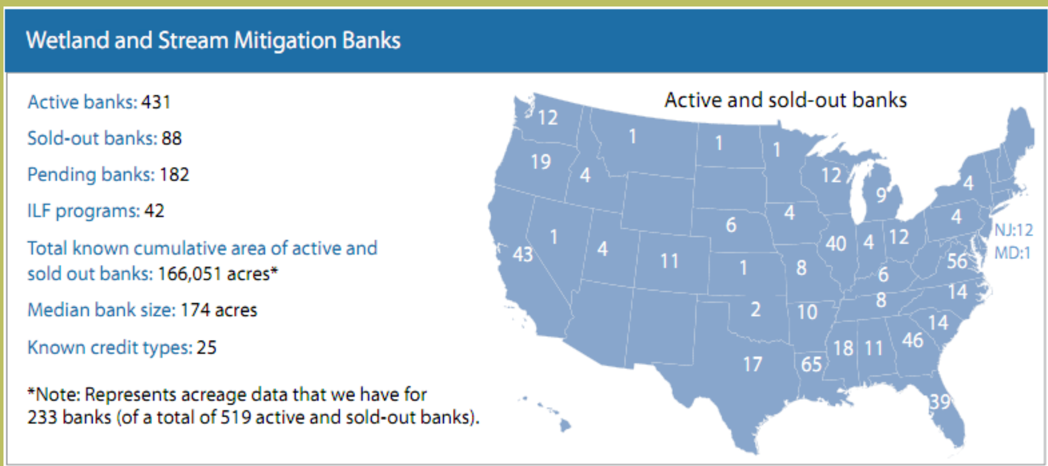
Source: Madsen et al. (2010)

Figure 1. Credit pricing



Source: Madsen et al. (2010)

Figure 2. Mitigation Banks



Source: Madsen et al. (2010)

3 Actors involved

Habitat banking and biodiversity offsets are being applied for a wide variety of situations, involving several actors with different institutional roles. Usually, these schemes entail three main actors - buyers, sellers and regulators, as illustrated in figure 3, which are fundamental for the design and implementation of the instrument. As in any market, an essential element is that prices be determined by the collective supply and collective demand of individuals willing to exchange goods and services (Carroll et al., 2008). A regulatory authority is required for a functioning permit scheme, in order to oversee this process and to monitor and enforce its implementation (Wissel and Wätzold, 2010).

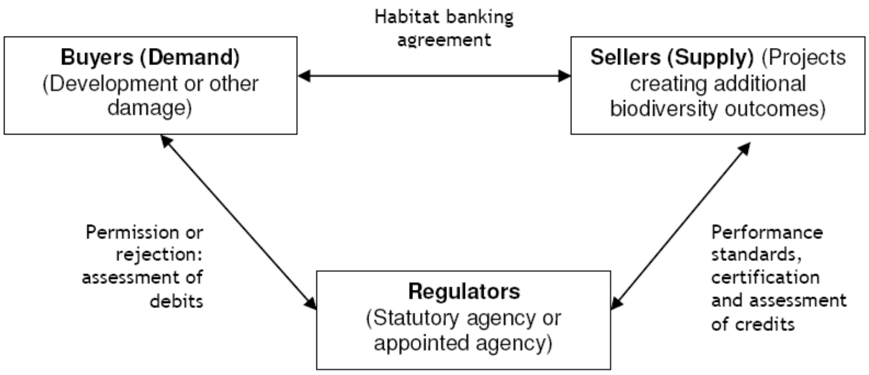


Figure 3. Typical key actors and roles in habitat banking systems
Source: eftec et al. (2010)

The institutional framework illustrated on figure 3 can change, for instance if the scheme includes an independent trust fund to allocate the funds received in lieu of biodiversity debits. This happens for example on the Australian Biobanking Scheme, and according to eftec et al. (2010) this arrangement has two main advantages: a) "the capability to enforce the purchase of credits according to strategic conservation priorities maximising the benefits of delivering no net loss"; and b) "the potential to reduce transaction costs and thereby enable compensation for minor impacts on widespread biodiversity, which would otherwise be unlikely to be covered by more complex compensation mechanisms".

The demand for credits may arise from different types of stakeholders and it can be determined by various reasons. Typically, demand is generated by developers such as government agencies, private firms, corporations or other organisations, whose activities need to meet regulatory obligations or comply with corporate social responsibility objectives (Bovarnick et al., 2010; eftec et al., 2010). The rational for their engagement is that private developers (e.g. real estate developers) can reduce costs by buying credits rather than offsetting their impacts themselves (Carroll et al., 2008). Philanthropy of individuals or NGO can also generate demand for credits, although they generally keep them, thereby enhancing the ecological value in a region (Bovarnick et al., 2010; Wissel and Wätzold, 2010).

According to eftec et al. (2010) the suppliers of credits will be "those with suitable land for whom creating and selling credits offers profit opportunities". Supply, in great majority, comes from private landowners, such as farmers and forest owners, and land managers. Government bodies and conservation groups in possession of land are other potential suppliers of credits (Wissel and Wätzold, 2010). Developers can also create credits, although this is not a very common option.

The credits trading process must be regulated, and in habitat banking is particularly important to have an effective regulator, as market characteristics are far from competitive requirements (Carroll et al., 2008). This regulator, that should be either an environmental authority or a nature conservation organisation (eftec et al., 2010), has a critical role in several stages and processes of this scheme, namely, on its design, implementation, monitoring and enforcement. When the banking scheme is locally implemented, local governments have a key role in the market regulation.

Market regulation is essential for the functioning of the market and to ensure that the conservation target is met (Wissel and Wätzold, 2010). The regulator must carefully set up the legal basis of the instrument in a way that fits within the existing policies and laws, but the regulator also has to design the rules that determine which habitats are equivalent (destroyed vs created) and how to assess their value (Wissel and Wätzold, 2010). A different task will be to ensure the transparency of the scheme (e.g. full documentation of all aspects of debit and credit calculation must be available), the accessibility of data to all stakeholders, and guidelines for scheme participants to create certainty and minimise the costs and risks resulting from regulation (Organisation for Economic Co-operation and Development (OECD), 2004). Regulators also need to ensure that this is a effective instrument to compensate the damage caused to biodiversity, for that purpose it is necessary to enforce, monitor and audit the ecological, legal and financial requirements (eftec et al., 2010).

Despite the importance of the three actors previously described, trading schemes are still more complex and involve a series of other stakeholders, known as "third parties", whose engagement is crucial for success (eftec et al., 2010).

Bovarnick et al. (2010) have identified some of the employment opportunities created by habitat banking, grouped by employment type (see Table 2), including some actors that will be involved in the scheme by providing knowledge and/or services to the three main stakeholder groups (regulator, buyers and sellers). One example is the provision of scientific and market support services by NGOs or universities (e.g. technical support for habitat conservation and restoration, monitoring and evaluation services). Some authors also refer that some industry clusters have been specialised specifically in enhancing or restoring wetlands in order to sell wetland mitigation credits (Bayon, 2002 cited by Burgin, 2010).

Table 2. Employment opportunities on habitat banking

Employment type	Specific employment opportunities
Design, establishment and maintenance of habitat banks	Wetland conservation scientists, biodiversity conservation scientists, hydrological engineers, conservation wardens, landscape engineers, forestry professionals, habitat restoration experts, construction workers
Monitoring, evaluation and verification	Wetland conservation scientists, biodiversity conservation scientists, forestry professionals, habitat restoration experts
Legal support	Property lawyers, financial lawyers
Registry and administration	Market administrators, registry specialists, public administrators
Project finance & banking services	Investment bankers, venture capitalists, commercial bankers
Market information services	Market researchers, news and intelligence analysts
Fund creation and management	Investment fund managers, fund management consultants
Project technical support	Environmental consultants with knowledge of habitat and wetland restoration, NGO specialists, researchers

Source: Bovarnick et al. (2010)

Beyond those actors engaged through the creation of job opportunities there are other relevant stakeholders, such as the local communities whose agreement to the proposed land-use changes is a key aspect of the planning process (Carroll et al., 2008). Banks, insurers and other financial service providers are also stakeholders with increasingly important roles (e.g. to manage endowments capital in terms of risks). The involvement of banks in biodiversity offsets programmes and conservation banking could enable them and their clients to improve risk management, as well as to explore business opportunities presented by the mitigation hierarchy and offsets (PricewaterhouseCoopers, 2010).

Conservation brokers are a new actor that has recently entered the habitat banking schemes; they are relevant to facilitate the transaction of credits when the potential number of participants in the market becomes large. Basically, they identify landowners with suitable land for habitat banks and then assist them to put together and market their projects (eftec et al., 2010). An example of their role can be seen in the Bushbroker scheme in Victoria, Australia (Box 2). According to Treweek et al. (2009) their participation will help to gain market information on the availability of credits and their likely price. However, it may also limit the ability to strategically locate compensation measures in areas that will provide the best biodiversity observation outcomes.

Tradable Development Rights (TDR) schemes typically involve private landowners and a regulating authority (e.g. a regional planning board). TDR can be thought of as a way of encouraging the reduction of development in areas that should be saved ('sending zone') and increasing development in predicted growth areas ('receiving zones') (Pruetz, 2003) (see Figure 4). Sending zones could be established for several reasons (e.g. conservation potential, agricultural importance or scenic beauty). Similarly, receiving zones are chosen because there is an abundance of abandoned sites or infrastructure already in place to bolster further development.

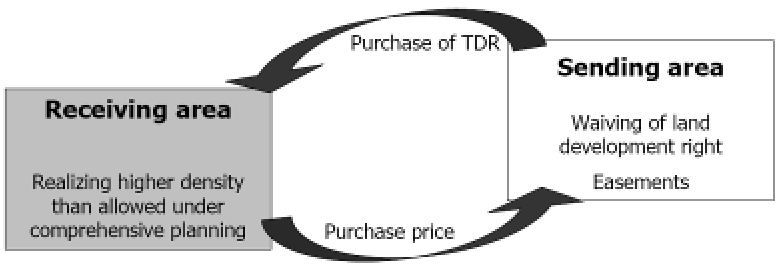


Figure 4. Schematic of a dual transfer district TDR-programme
Source: adapted from Süess and Gmünder (2005)

Landowners in sending zones are assigned TDR as compensation for the reduced property value of their sites due to restricted development options. Very often permanent easements are placed on these parcels, prohibiting their future development (Pruetz and Standridge, 2009). Landowners in receiving areas typically face a dual zoning: developers can choose to build at the baseline density (e.g. x dwelling units per hectare or to acquire TDR in order to achieve a higher, more profitable level of development (Machemer and Kaplowitz, 2002a) (e.g. y dwelling units per hectare). The price of the development right and thus the compensation received by landowners in sending zones is subject to demand and supply.

Besides trading of development rights, some jurisdictions in the US are using public money to directly buy out development rights from private landowners. The owners of farmland, open space and natural areas are given the option of recording a deed restriction on their property ensuring permanent preservation in return for selling their development rights (Pruetz, 2003: 82). These schemes are referred to as PDR – Purchase of Development Rights. Since their operations are constrained by the public budgets dedicated to this purpose, preservation success is significantly lower than in TDR-terms, judging only on quantitative terms, i.e. number of hectares preserved.

Although some stakeholders have not yet completely embraced the concepts of offsets, habitat banking and tradable permits, part of the environmental movement recognises them as a mechanism for companies to secure and maintain their license to operate, and a method for investors to minimise risks associated with impacts on biodiversity (Burgin, 2008). Fox and Nino-Murcia (2005) stress that offset projects rise above the antagonistic relationships usually associated with conservation and development, since they favour the establishment of collaborative partnerships between landowners, biologists, consultants, planners, and developers.

Box 2. Victoria's "Bushbroker" Scheme

In the State of Victoria (Australia), one of the options for developers to comply with "native vegetation offsets" required under planning law is to purchase biodiversity credits through a government regulated trading system. The Bushbroker programme was initiated in 2006, with the goal of halting or reversing loss of native vegetation, beyond that, under the Native Vegetation Act, developers in the State of Victoria are required to achieve the no net loss goal (Department of Sustainability and Environment (DSE), 2010).

The trading units are the Native Vegetation Credits, where a native vegetation credit is a gain in the quality or extent of native vegetation. The credit traded can be defined by three possible units: vegetation or habitat; "large old trees" (LOTs); and "new recruits" (i.e., tree planting) (Department of Sustainability and Environment (DSE), 2010). Offset supply has generally been from agricultural landowners and credit demand generally comes from road building, housing development, water supply pipeline development, and landholder vegetation clearance.

Using data from the State of Biodiversity Markets Report (2010) it is possible to track the programme's implementation in the period of 2006 to 2009, particularly the number of transactions and the price of credits. For the period from May 2006 to November 2009, the cumulative habitat offsets under this programme is of around 522 hectares, with an estimated monetary volume of offsets of AUS$ 11,358,720 (Madsen et al., 2010), approximately 8.4 million Euros.

Table 3. Transactions under the Bushbroker programme

Transactions	
2007/2008*	**2008/2009***
35 offset transactions	63 offset transactions
49.2 habitat hectares	11.23 habitat hectares
264 'large old trees'	166 'large old trees'
6,959 'new recruits'	13,140 'new recruits'

Source: Madsen et al. (2010)

Table 4. Prices of credits for the period of 2006 to 2009

Credit pricing for habitat hectares alone or habitat hectares + Large Old Trees (LOTs) between May 2006 -November 2009**				
Bioregion	**Average price per habitat hectare*** **	**Habitat hectare price range********	**Total number of habitat hectares**	**Estimated AUS$ volume of offsets**
Goldfields	$39,000	$17,000 - $86,000	35.8	$1,396,200
Victorian Volcanic Plain	$167,000	$36,000 - $293,000	49.28	$8,229,760
Gippsland Plain	$156,000	$85,000 - $250,000	4.91	$765,960
Other bioregions	$80,000	$16,000 - $157,000	6.76	$540,800
Credit pricing for LOT credits between May 2006 - November 2009**				
Bioregion	**Average price per habitat hectare*** **	**Habitat hectare price range********	**Total number of habitat hectares**	**Estimated AUS$ volume of offsets**
All bioregions	$1,000	$300 - $2,900	426	$426,000

Source: Madsen et al. (2010)

4 Baseline

A mitigation hierarchy, establishing that appropriate measures should be identified and taken to avoid and reduce the potential impacts of a development, and where necessary to compensate for residual impacts, is generally accepted in biodiversity conservation policies (Crowe and ten Kate, 2010; PricewaterhouseCoopers, 2010; eftec et al., 2010; Madsen et al., 2010). This hierarchy forms the basis for compensatory mitigation, "wherein an unavoidable impact to the environment is offset by the restoration, protection or conservation of a similar area or environmental attributes elsewhere" (Bovarnick et al., 2010). A baseline hierarchy is defined by Business and Biodiversity Offsets Programme (2009), eftec et al. (2010) and Bovarnick et al. (2010), considering the following concepts:

Avoidance: "measures taken to avoid creating impacts from the outset, such as careful spatial or temporal placement of elements of infrastructure, in order to completely avoid impacts on certain components of biodiversity. This results in a change to a 'business as usual' approach".

Minimisation/reduction: "measures taken to reduce the duration, intensity and/or extent of impacts that cannot be completely avoided, as far as it is practically feasible".

Rehabilitation: "measures taken to rehabilitate degraded ecosystems/habitats or restore cleared ecosystems/habitats following exposure to impacts that cannot be completely avoided and/or minimised".

Offset or compensation measures: "offsite measures taken to compensate for any residual significant, adverse impacts that cannot be avoided, minimised and/or rehabilitated or restored, in order to achieve No Net Loss or a net gain of biodiversity. Offsets can take the form of positive management interventions such as restoration of degraded habitat, arrested degradation".

The generally adopted hierarchy clearly sets offsetting as the last option in managing biodiversity conservation. For example, in the EU Habitats Directive (92/43/EEC), under the Article 6, assessment process compensation is a measure of last resort if there are no alternatives to the plan or project and if there are demonstrable imperative reasons of overriding public interest (Gillespie and Hill, 2007). However, offsets are an available option for businesses and governments to mitigate their impacts on biodiversity, but they must be additional to other measures that are in place to avoid, or minimise, environmental damage (ten Kate et al., 2004; Burgin, 2008).

Property rights are important for this type of instrument because they grant restricted access to a resource, so that the property owner is secure that today's investments will generate tomorrow's returns, for that matter they must be created and well defined (Organisation for Economic Co-operation and Development (OECD), 2004). Defining property rights varies from country to country. It can be done through environment indicators or indexes, or in countries such as the US, as a bundle of rights associated with land, such as lease, use, and development rights (eftec et al., 2010). In the Biobanking scheme (Australia), for example, the allocation of property rights is made through the signing of legally binding contracts with land managers (sellers), and those buying credits are committing to secure the conservation of biodiversity in perpetuity (Department of Environment and Conservation (DEC), 2005).

Most TDR programmes are implemented on top of some type of existing zoning system that establishes maximum density limits in different parts of an urban region (McConnell and Walls, 2009). Thus, the baseline is defined by the existing planning regulation for land development. In some existing programmes, there is a parallel reduction of the baseline zoning in the receiving area. In this case, developers have to purchase development rights even when they wish to build at the previously allowed density. This design feature further strengthens TDR demand and thereby TDR-prices, and is thus profitable for sending zone landowners.

5 Monitoring and evaluation

A major issue with the management of offsets, habitat banking and tradable permits, as with other environmental policy instruments, is compliance (Burgin, 2008; ten Kate et al., 2004). Effective monitoring and verification of the biodiversity impacts are crucial to ensure the long-term environmental integrity of any offsets and habitat banking schemes. However, the costs and benefits of monitoring have to be carefully balanced (Bovarnick et al., 2010).

According to eftec et al. (2010), monitoring is necessary to:

- ensure legal compliance, with respect to actions/processes, biodiversity impacts and where possible their additionality (by comparison with sites over time);
- facilitate adaptive management of individual projects;
- provide scientific feedback on the effectiveness and costs of particular measures to authorities responsible for schemes;
- provide feedback to other stakeholders, e.g. conservation organisations and local communities; and inform policy development.

Monitoring must be carried out by the regulator or accredited third parties, while its costs are typically borne by the developer. Transparency in verification processes can increase buyer confidence (Bovarnick et al., 2010). Biodiversity monitoring has several stages; the first must start with extensive baseline surveys and the development of several agreements. On-going monitoring of the process and post-implementation monitoring must unfailingly establish whether has met or not its overall objectives of no-net-loss (Burgin, 2010) by providing sufficient biodiversity benefits to compensate for observed or expected losses. To do that, monitoring efforts must focus on the ecological performance criteria adopted during the agreement phase.

Treweek (1999, cited by Gillespie and Hill, 2007) stated that mitigation schemes were considered inadequate because no monitoring was proposed or undertaken, there was no follow-up once the development started. Recent reviews of mitigation banking schemes have shown high rates of non-compliance with agreed conditions (Gibbons and Lindenmayer, 2007).

Effective monitoring is also an important prerequisite for the success of TDR-markets. On the one hand, sending zone sites, once TDR are sold, have to be permanently protected to eventually achieve conservation success. On the other hand, the stability of fixing sending zone is essential – if landowners expect that the imposed restrictions on development will be eased in the future, willingness to sell TDR will be diminished.

In the reviewed case studies, several schemes include mandatory monitoring and give a high priority to ensure that it is effective. However, it appears to be ineffective in establishing the success of reaching the ecological goals.

6 Policy analysis

This section presents a concise synthesis of this instrument performance in relation to several selected evaluation criteria – effectiveness, cost-effectiveness, social and distributional impacts, as well as institutional requirements based on a sample of twelve case studies reported in the literature.

6.1 Effectiveness for biodiversity conservation

There is an abundance of literature dealing with the search for and the explanation of success factors for TDR-programmes. Pruetz and Standridge (2009) are able to identify 20 publications that mention 55 individual success factors (Bredin, 1998; Costonis, 1974; Coughlin and Keene, 1981; Farmland Information Center, 1997; Field and Jon, 1975; Glickfeld, 1990; Heeter, 1975; Juergensmeyer et al., 1998; Kaplowitz et al., 2008; Lane, 1998; Machemer and Kaplowitz, 2002a; Machemer and Kaplowitz, 2002b; McConnell et al., 2007; Meck, 2002; Merriam, 1978; Pizor, 1986; Roddewig and Inghram, 1987 ; Stinson, 1996; Strong, 1998; Tripp and Dudek, 1989; Walls and McConnell, 2007). Pruetz and Standridge (2009) assessed ten factors that were cited in five or more articles. Their analysis was based on the assumption that success of a programme can be described by the amount of land preserved. Five factors were deemed as important for programme success, of which the first two were found to be most critical:

1. Developers must want the additional development only available through TDR;
2. receiving areas must be customised to work within the physical, political and market characteristics of the community;
3. development possibilities on sending sites must be strictly limited;
4. developers are offered only few alternative ways of gaining additional development potential other than TDR;
5. programmes offer market incentives like transfer ratios and conversion factors designed to produce TDR prices that adequately compensate sending area landowners, yet are affordable to receiving area developers.

Some communities have begun to learn these lessons. For example, the Pinelands Development Corporation, which runs the Pinelands TDR programme in southern New Jersey, now prohibits municipalities from granting ‘free’ density above the baseline through variances to zoning rules – any additional density must be purchased with TDRs (factor 4) (McConnell and Walls, 2009).

Among the more than 140 TDR-programmes in place in the USA, programme designs differ greatly, and the results vary largely (Walls and McConnell, 2007). Some programmes are huge success stories, in terms of trading activity (i.e. programme participation) and in hectares of open space and prime farmlands protected (see Table 5). Notably, the TDR-programme in Montgomery County, Maryland is held to be one of the most successful schemes. By 2008 it had preserved over 50,000 acres of prime agricultural land and open space in the densely-developed Washington, DC/Baltimore corridor by transferring more than 8,000 development rights, accounting for 75 per cent of all preserved agricultural land in the county (Pruetz and Standridge, 2009).

According to the State of Biodiversity Markets Report (2010), there are 39 trading programmes around the world, and another 25 in various stages of development or investigation. The global market size is $1.8-$2.9 billion at minimum, and likely more, as 80% of existing programmes are not transparent enough to estimate their market size. The conservation impact of this market includes at least 86,000 hectares per annum of land under some sort of conservation management or permanent legal protection.

In the surveyed literature it is not possible to find a definitive and agreed conclusion about the environmental effectiveness of habitat banking and offsets, in the sense that the instrument enables to achieve the environmental goals. While in some case studies the reported results are good and the main goals are completely fulfilled, in others only partially, and there are also some examples of related failures.

Table 5. The 20 US TDR-programmes that have preserved the largest acreage

TDR-program location	Year of adaption	Hectares preserved as of 2008	Average hectare preserved per year
King County, WA	1998	37.028	3.703
New Jersey Pinelands, NJ	1981	22.623	838
Montgomerey County, MD	1980	20.974	749
Palm Beach County, FL	1993	14.164	944
Collier County, FL	2002	12.707	2.118
Calvert County, MD	1978	5.366	171
Queen Anne´s County, MD	1987	4.523	51
Sarasota County, FL	1982	3.318	369
Pitkin County, CO	1994	2.611	187
Boulder County, CO	1989	3.388	126
San Luis Obispo County, Ca	1996	2.211	184
Blue Earth County, MN	1970	2.169	87
Howard County, MD	1992	1.831	115
Miami/Dade County, FL	1981	1.677	62
Payette County, ID	1990	1.677	93
Charles County, MD	1992	1.655	104
Rice County, MN	2004	1.558	390
Douglas County, NV	1996	1.509	126
Collier County, FL	2004	1.396	349
Chesterfield Township, NJ	1998	919	92

Source: Pruetz and Standridge (2009)

The environmental goals vary for the reported case studies, but it is possible to identify some of the main environmental benefits linked with habitat banking:

- It is usually part of a large conservation strategy, therefore resulting on the establishment of larger reserves and greater connectivity of habitats, due to a strategic and selective placement of compensation measures (e.g. to link up, increase the size of, or buffer Natura 2000 sites) that results in a reduction of habitat fragmentation;
- Its implementation in larger scales increases the potential for increased positive impacts and long-term viability of the conservation measures;
- It creates the option to trade up measures to address higher conservation priorities;
- It raises the opportunity to efficiently address cumulative impacts from small-scale or low impact developments for which there is no legal requirement for compensation;
- It reduces the temporal loss of habitat;
- It is a more effective, and in some cases ex-ante (and therefore more reliable), delivery of existing biodiversity policy objectives and of compensation requirements.

The US wetland mitigation scheme results are a strong argument in favour of this instrument, having contributing to the effective creation of thousands of acres of wetlands and protected sites “that would not have existed had the law not required developers to offset their impacts on wetlands in this way” (ten

Kate et al., 2004). This statement is based on the fact that developers are often obliged to offset (e.g., protect, restore) larger areas of wetland than the ones that have been lost due to development (Burgin, 2010).

Despite the reported substantial gains, mitigation efforts are often criticised for failing their goals, and instruments like habitat banking face several difficulties and risks. The main problems include the risk of allowing development too damaging, as well as the failure to deliver the "no net loss" goal and the equivalence in terms of the impacted biodiversity.

6.1.1 Allowing development too damaging

Offsetting shall be the last option in dealing with biodiversity conservation. This is a major concern regarding this instrument because there is the risk of allowing development too damaging to the environment, what has been termed by eftec et al. (2010), as "licence to trash", resulting on the introduction of perverse incentives. This happens when compensation measures are easier and cheaper, leading to the approval of destructive development that would not have been permitted in the absence of compensation options. To avoid this problem it is necessary to effectively apply and enforce the mitigation hierarchy. Although regulators must ensure that laws do not change because of habitat banking, society also has an important role, defining what constitutes an acceptable trade-off between avoiding and mitigating impacts on-site, versus off-site compensation through offsets or habitat banking (Carroll et al., 2008).

6.1.2 No-net-loss goal

One of the goals of offsets is to achieve no net loss of biodiversity or a net gain of biodiversity. The principle of no-net-loss is intended to prevent the loss of ecosystems and their functionality (Bovarnick et al., 2010). This implies that the species or habitat must be created, elsewhere within the ecosystem or species range, typically on a per-area basis to compensate the loss that will occur on the original area due to development (Burgin, 2008; eftec et al., 2010).

There are serious questions raised about the success of this goal. Burgin (2010) identifies several reviews about the US wetland mitigation (e.g., National Research Council (NRC), 2001; Turner et al., 2001; Kihslinger, 2008) that conclude it failed to achieve the no-net-loss goal, and suggested that the mitigation areas are not near to compensate the size of the affected area. Kentula et al. (1992, cited by Burgin, 2008) also suggests that "the US Army Corps of Engineers, the overseers of the wetland developments, have failed to keep adequate records to enable the assessment of whether 'no net loss' has been achieved". Turner et al. (2001) suggested that approximately 80% of wetlands built for mitigation did not become fully functional. Hallwood (2007) also points out that only about 25% of the mitigation wetland projects were ecologically successful in the sense that they had or would probably become serviceable wetlands of the type permitted. Beyond the critiques to the achievement of the no net loss goals, Carruthers and Paton (2005) state that the concept of a trade between areas does also do not result in a net gain for biodiversity. There has been, however, a movement towards net-positive policies that seek to ensure environmental gains (Burgin, 2008).

6.1.3 Equivalence

According to Strange et al. (2002), ecological equivalence refers to "the capacity of a restored, created, or enhanced habitat to reproduce the ecological structures and functions provided by a resource before injury". In trading schemes such as habitat banking, it is essential to ensure the equivalence between the biodiversity values lost (e.g. habitats, species) and the restored/created values, in order to, at least maintain the overall conservation value (Wissel and Wätzold, 2010). Without establishing the equivalence between gains and losses the credit provision remains a type of compensation and not a true biodiversity offset that has achieved no net loss.

Determining equivalence between the type of damage and offsets or habitat banking credits is difficult to define and achieve (eftec et al., 2010). Morris et al. (2006) argue that "some elements of the natural environment can clearly be restored, created or re-created while there are others for which there is limited evidence of re-creatability". According to several authors (e.g. Salzman and Ruhl, 2000; Ring et al., 2010a; Wissel and Wätzold, 2010) the problems to establish equivalence arise mainly due to three dimensions: type (restored and destroyed habitat provide different functional values), space (configuration and connectivity of sites matters) and time (restoration of habitat requires time, leading to increased vulnerability).

Type "refers to the kind of habitat that is present in the destroyed and restored locations, such as forest, grassland and wetland" (Ring et al., 2010a). To date offsets tend to be "like-for-like", largely due to the difficulties of developing accurate and cost effective measurements of baseline biodiversity (Burgin, 2008). This concept requires that permits correspond to the same environmental characteristics (e.g. species) as the ones degraded or destroyed elsewhere (Bovarnick et al., 2010). Allowing trade between different habitat types may increase cost-effectiveness, however it can lead to uncertainty about the future value of land, and the complex regulation can increase transaction costs (Wissel and Wätzold, 2010). The persistence of biodiversity in restored/created is highly sensitive both to the spatial and temporal allocation of the habitat (cited by Drechsler and Hartig, 2011; Hanski, 1999; Roy et al., 2004; Wilson et al., 2007; With and King, 2004).

Space "concerns the spatial configuration of habitat at the landscape scale, which changes as a result of the permit market" (Ring et al., 2010a). The size and ecological connectivity of habitats are very relevant, especially for species: connected habitat sites are more valuable than isolated ones (Hanski, 1998 cited by Wissel and Wätzold, 2010). So, wherever possible, restored sites must be adjacent to the destroyed site to properly offset impacts (Morris et al., 2006), and need to be large enough on their own or in connectivity with adjacent protected habitats in order to ensure a size genetically secure of the populations (Bonnie, 1999). Neglecting spatial dimension of trading rules will likely be less cost-effective than with rules that include it (Hartig and Drechsler, 2009). However, such rules (more complex) will probably increase transaction costs (Ring et al., 2010a).

Time "concerns the continuous availability of habitat" (Ring et al., 2010a). In trading schemes this dimension raises criticism because of the time lag between the destruction of a habitat and its full restoration, there is an instant loss of biodiversity traded for slow gain (eftec et al., 2010). It is then critical to ensure that restoration actually takes place. To do so, Wissel and Wätzold (2010) suggest several strategies: one is to "keep the payment for the permit or part of it with a trustee until the new habitat is completely restored", other is to "allow the trade only after restoration has been successful". The use of intermediate milestones can also help to ensure that the processes are on track (Edgar et al., 2005; Gibson, 1995 cited by Morris et al., 2006). However, these strategies might create uncertainty (on the supply side) about the future permit price and can limit market activity and reduce cost saving potentials (Wissel and Wätzold, 2010).

Burgin (2008) identifies some important weaknesses of these schemes, including the: a) difficulties to clearly define biodiversity, b) limitations to calculate biodiversity values with existing science, c) management and compliance problems, and d) an overall lack of resources for implementation and long-term monitoring.

Habitat banking and offsets schemes function best if there is an effective measurement of biodiversity values gained and lost, in a way that impacts are clearly quantified and compared, in order to allow stakeholders and authorities to recognise the outcomes (eftec et al., 2010; Wissel and Wätzold, 2010). There are, however, several difficulties to accomplish this. One is the lack of methodologies and tools to quantify the impacts and benefits on biodiversity of the proposed offsets (Burgin, 2008). When compared with other areas of science, such as physics or mathematics, the available science for estimating the extent of restoration required to achieve equivalent ecological services is much more imprecise and complex (eftec et al., 2010).

Depending on the conservation target, data availability might also be a problem. Conserving ecological communities is relatively simple to evaluate (e.g. air photos, GIS classification, species lists) when compared to assessing endangered species or ecosystem services (ten Kate et al., 2004). Where data are absent, Burgin (2008) suggests a combination of the precautionary principle, ecological and economic valuation to support decision-making.

This leads to another problem: in order to measure the ecological value of a destroyed and restored site an exchange unit is required (Wissel and Wätzold, 2010). Unlike greenhouse gas trading schemes where carbon credits that are based on measurement of a single quantifiable unit (a ton of carbon dioxide), biodiversity values are complex to measure, especially ecosystem service roles, and currency units may not be easily fungible, limiting market liquidity (Salzman and Ruhl, 2002; Bovarnick et al., 2010). The use of a single unit (e.g. area), allows easier measurement but it is only a rough measure of conservation values because it ignores innumerous factors, such as habitat connectivity (Wissel and Wätzold, 2010). On the other hand, the use of a more accurate exchange unit will increase complexity and can eventually be very difficult or impossible to measure (Bonnie, 1999).

Analysing the selected case studies, it is possible to conclude that, even in the same country, different methods are used to assess biodiversity, adopting different indicators and different units. So, it is not possible to guarantee the comparability between case studies, neither to assess the global performance of this instrument.

The US Wetland Banking scheme and the Conservation Banking scheme in USA have a very similar regulatory approach; however the units used are completely different. California developed the first formal policy on conservation banking and established the first conservation bank in San Diego County, California. It took the concept of wetland mitigation banking and applied it to endangered species. The regulatory requirements result from the Endangered Species Act (ESA), which are the drivers of a set of habitat banks known commonly as “endangered species conservation banks” or simply “conservation banks”. ESA under Section 7 or Section 10, requires project proponents to obtain a permit if their actions will likely jeopardise the continued existence of an endangered and threatened species. This difference makes impossible to compare both schemes in terms of the ecological outcomes or credit value.

Beyond those previous problems, trading schemes have difficulties to ensure additionality of the projects (Madsen et al., 2010). In some cases, not all credits sold are in fact additional benefits to biodiversity, outcomes sold as credits would occur anyway, for example due to the natural evolution of the habitat or due to management actions that are already in place (eftec et al., 2010). Burgin (2010) stated that “there have been over 16,000 hectares of conservation banks developed under US mitigation schemes, but 75% or more would probably have been developed even without legislation to mitigate loss”. An example is the Stillwater Plain Conservation area (US). This area was not in danger of immediate loss because it was uneconomic to develop the area for housing (Bayon, 2002), so the trading of wetlands under threat elsewhere with credits on this land cannot be considered as an additional benefit for biodiversity conservation (Burgin, 2010).

There are also several drawbacks to the effectiveness of TDR-programmes. Since most schemes work on a voluntary basis, development rights are firstly sold by those sites that are most unlikely to become developed anyhow; thereby revealing an adverse-selection of the sending zone-sites. Moreover, as transfer of rights is usually done on a simple 'hectare per hectare' basis, differences in the conservation potential of sending site-parcels are neglected.

In Purchase of Development Rights (PDR)-schemes, in contrast, there is typically an assessment of the different sites and their ecological features, importance and underlying threat of development. Therefore, these programmes very often achieve higher conservation results by preserving less acreage (Brabec and Smith, 2002; Lynch and Musser, 2001). Many TDR-programmes are facilitated by a PDR-sub-programme that is directed towards acquiring development rights from ecologically important sites, e.g. buffer zones or corridor parcels that are highly threatened by development.

6.2 Cost-effectiveness

The costs and benefits associated with the development and implementation of these instruments have significant variations, even within the same country, and, for this reason the real evaluation of costs and benefits requires a case-by-case approach (Crowe and ten Kate, 2010). However, some general conclusions are presented in recent publications.

Habitat banking and offsets have several economic benefits, including, like in other trading schemes, the reduction of transaction costs, both of regulation and pairing up buyers and sellers (Carroll et al., 2008). Burgin (2008) also defends that in certain areas, e.g. less developed areas or with lower opportunity costs, this instrument "can achieve better and more cost-effective conservation outcomes than other options". According to eftec et al. (2010), "a theoretical and empirical economic analysis, shows that habitat banking schemes compare favourably to other market based policy instruments for biodiversity conservation. This favourable comparison is contingent on it being possible to design an efficient system, which balances regulatory controls of risks with freedom for the market to operate and capture efficiency gains".

Another major benefit of this instrument is the establishment of a market for property rights of biodiversity resources that provides an incentive for landowners to "designate their land in such a way that a cost-effective allocation of land-use types emerges" (Wissel and Wätzold, 2010). This can be an extremely important tool to engage landowners in biodiversity conservation, by replacing a liability for an opportunity (Carroll et al., 2008).

A further advantage lies in the fact that mitigation banking offers greater fixed certainty to developers (Wissel and Wätzold, 2010). Knowing the predicted outcome for the mitigation project increases confidence levels of the developers. By contrast, developers "dislike uncertainty that can lead to escalating costs and no exit strategy" (Gillespie and Hill, 2007). This is a very relevant aspect when comparing mandatory offsets and voluntary offsets. The advantage of regulatory approaches is that developers are clear about the nature, scope, and sometimes even the cost, of their obligations. Beyond that, developers know that these laws apply equally to all competitors, so that engaging in offsets is not a competitive disadvantage (ten Kate et al., 2004). Developers prefer regulatory interventions to trigger conservation banking and create markets, existing some scepticism about biodiversity offsets on a voluntary basis (Crowe and ten Kate, 2010). There is also the idea that without regulation to back it up, offsetting activity is likely to be unpredictable, and the first thing to go in hard times (ten Kate et al., 2004). This is very clear in the United States where regulatory frameworks such as the Clean Water Act and the Endangered Species Act are much more powerful incentives for offsetting behaviour than voluntary approaches.

Furthermore, as the habitat bankers will already have identified a number of sites where restoration is needed, measures can be very rapidly undertaken (thus minimising interim losses for services), and presumably, at a cost effective way. At the same time, the market would lead to a competition among companies that will encourage the most cost-effective options to come forward and provide compensation (eftec et al., 2010).

However, the market-based schemes are very dependent on their capacity to generate sufficient supply and demand for the credits. The supply of credits is mainly dependent on the availability of appropriate land, and the situation varies according to the type and location of the habitats. In habitats with high land-use values and properties with high prices (typically coastal areas) the supply side may be a constraint, while in other habitats with less economic productivity it is easier to find a sufficient supply of land (eftec et al., 2010). The demand for credits is especially significant because it has a huge influence on the potential supply of credits. Credits suppliers face set up costs and opportunity costs that need to be overcome with high demand for credits, in order to successfully implement the market (Carroll et al., 2008). Market demand has to be big enough to incentive the adoption of the conservation actions, which is closely related to the opportunity cost associated with alternative land uses. There are significant differences between land values range in areas with high development potential, e.g. urban, compared to rural farmland or forest land (Bonnie, 1999), which can lead to changes in the geographic distribution of mitigation banks.

A regional size market with a larger number of participants typically leads to reduced information costs, higher opportunity-cost differences and, hence, higher trading activities (Wissel and Wätzold, 2010). In summary, the availability of appropriate land for habitat banking is dependent on the relative price for different uses. In this respect, the market system of habitat banking has an advantage of providing price signals that can promote an efficient allocation of land between different uses. If biodiversity compensation is required by law, the market gives an incentive for credits to be priced at a level that is sufficient to secure appropriate land for their delivery (Carroll et al., 2008).

Not all TDR programmes operate in exactly the same way, but they all include the feature that density is transferred from one area to another. By allowing for these voluntary transfers, the programmes have the potential to improve efficiency compared to a 'command-and control' system of zoning alone in which density limits are assigned uniformly across multiple property owners (McConnell and Walls, 2009). In fact, modelling studies already done in the 1980s have shown that these types of tradable systems have efficiency advantages over a zoning-only policy (Carpenter and Heffley, 1982; Mills, 1980). Very recently, the efficiency of tradable permits for biodiversity conservation is analysed from an ecological-economic-modelling perspective (Drechsler and Hartig, 2011).

From a local government's perspective, TDR offer a huge advantage, as land is preserved without expenditure of government money. Walls and McConnell (2007) estimate that preserving the 48,000 acres of land in Montgomery County, Maryland would have cost the county approximately US$ 68 million if done through purchase of development rights. Most communities do not have the resources to preserve the amount of land they would like to preserve through a PDR or land purchase programme; thus TDR provide an attractive alternative. Although this is not directly a feature of instrument efficiency, it is a clear advantage of trading systems above other compensation measures, like tax-financed payments for ecosystem services (PES).

According to the State of Biodiversity Markets Report (2010), there are 39 trading programmes around the world, and another 25 in various stages of development or research. The global market size is US$ 1.8-2.9 billion at minimum, and likely more, as 80% of existing programmes are not transparent enough to estimate their market size. The conservation impact of this market includes at least 86,000 hectares per annum of land under some sort of conservation management or permanent legal protection.

Implementing offsets, habitat banking and trading schemes has also some risks and disadvantages linked with economic aspects. A main issue is the potential resistance that developers or other stakeholders can have regarding offsets and banking. The instrument needs to be able to attract support, using arguments such as its role to internalise environmental costs in an efficient way (eftec et al., 2010). However, developers with lower experience and expertise in habitat restoration are likely to have higher costs than more experimented developers (Hallwood, 2007).

According to Wissel and Wätzold (2010), the instrument design may also have a negative impact on several other market-related factors, namely, reduced frequency of transactions due to low demand for tradable permits (typically in regions with little economic growth), additional regulation may restrict trading opportunities, equal opportunity costs reduces the incentive to trade. Complex administrative procedures can lead to high transaction costs, thus reducing also the market activity. In the trading schemes, typically, there is a trade-off between assessing a sending site's characteristics in order to adjust permit price (e.g. in PDR-programmes or by assigning a special transfer ratio) and the transaction costs associated with a TDR-scheme. Complex trading schemes, involving individual assessment of sending sites, or single trade were found to have substantially lower numbers of transactions, programme participation and hence conservation effect (Machemer et al., 1999; Walls and McConnell, 2007; Woodward, 2003).

A key constraint regarding the potential supply of credits is the viability to restore different types of biodiversity. Since restoration is associated with both opportunity and restoration costs, biodiversity resources that take longer to restore are very costly, thus increasing the investment risk for potential suppliers (Drechsler and Hartig, 2011). Long time-scales means credit suppliers will take longer to deliver, leading to increased monitoring and management costs (Bean et al., 2008). The time necessary to restore habitats can be long, as is the case to regenerate some ecosystems, e.g. forests require decades or even centuries of growth (Drechsler and Hartig, 2011; Wissel and Wätzold, 2010), while in other cases, such as some wetlands, may take just a few years (Morris et al., 2006). For markets timescales over 50 years are unlikely to be commercially feasible; generally timescales of up to 10 years are more likely. Therefore, for this type of instruments the habitats of greatest relevance are usually those that can be restored over shorter timescales (eftec et al., 2010). This balance between environmental effectiveness and economic attractiveness is one of the key factors for the success of this instrument (Carroll et al., 2008).

The risk and costs of monitoring and management measures are leading to changes in the geographical distribution of the banks, namely by a transfer from coastal areas to rural areas due to land prices (Environmental Protection Agency (EPA), 2009). As an example, in the US, the depletion of remnant wetland habitat in urban areas is being offset in rural landscapes (Ruhl and Salzam, 2006 cited by Burgin, 2010). This means that the destruction of habitats in coastal areas (generally with more development) is being compensated on inland areas, in completely different habitats.

6.3 Social impacts

Offsets, habitat banking and tradable permits have both positive and negative social impacts. On the one hand, negative impacts can be caused to indigenous people and landowners who necessarily lose some cultural and traditional practices, especially regarding land management and farming or grazing practices. The need to integrate conservation measures in this type of activities leads, in the majority of the cases, to changes in the way these people manage their land. For that matter, new technical skills and knowledge are needed for landowners to do this transition from traditional practices to more suitable land management procedures (Hallwood, 2007). This can also be seen as a positive impact, as it means an increase in education and available information for these people, and also the engagement and empowerment of these stakeholders in conservation efforts.

Another positive social impact is the creation of new sources of income to landowners, through the transaction of credits but also due to the "business and job creation potential from the establishment, maintenance and monitoring of habitat banks" (Bovarnick et al., 2010), helping local communities to enhance quality of life standards (Crowe and ten Kate, 2010). These markets could contribute directly to achieve poverty alleviation goals especially in low-income rural communities, generally with low levels of employment and economic development. Beyond job creation, local communities can also benefit from another source of income: "community-based habitat banks" can also generate income for community development programmes and alternative livelihoods projects (Bovarnick et al., 2010). A positive side effect of this impact is that the creation of new sources of income for local communities can reduce subsidies from state or local level, allowing those resources to be spent on other needs of the communities.

Relevant stakeholders need to be involved right from the very beginning of the designing process to ensure community support for the project development plans and its successful implementation. Securing the support of local stakeholders, through participatory processes is thus essential for the success of this instrument. Stakeholders' involvement is important to prevent them to engage in activities that reduce the function of the credit site's habitat (e.g. illegal dumping, improper habitat use) and for them to understand the purpose and goals for the site. For example, site management plans should include stakeholder involvement from early stages.

On a very general level, TDR-schemes are deemed to increase social justice of zoning regulation, as development restrictions for property owners in sending zones are compensated by the return on sale of TDR, whereas developers in receiving zones have to pay for additional development exceeding prior existing legal limits. However, social impacts can be looked at in a much broader sense.

Building on the framework elaborated by Jacobs (1995), Cohen and Preuss (2002) critically assess the social equity issues in the Montgomery County TDR-programme. While the authors recognise the programme capacity to achieve 'intergenerational equity', understood as the programme's long-term effectiveness in preserving farmland, and 'tenure equity for new farmers', referring to the programme's capacity to ensure the availability of land affordable for new farmers, they found the programme to be performing less well on other criteria. 'Tenure equity for current landowners', referring to the programme's capacity for assisting farmland owners to get a fair economic return on their land, is deemed to be violated as there are no premiums to permit price for selling off development rights of ecologically critical parcels or contiguous blocks of land or from sites under heavy development pressure, and thus high opportunity costs. Another critique is placed on 'receiving area resident outcome equity', i.e. how receiving zone-residents receive benefits in exchange for accepting residential densities higher than allowed in base zoning, as often receiving sites have not got any additional support to accommodate additional density of development, e.g. for infrastructure provision. Lastly, 'process equity', consisting of two components: (a) opportunities for participation in TDR policy-making and programme development; and (b) promises kept to sending area landowners and receiving area residents are appraised as given.

6.4 Institutional context and legal requirements

To successfully implement this type of instrument a strong involvement of the public sector is necessary. Governments have an essential role on setting up the standards to instrument design, and also on regulating and monitoring its implementation. Lack of regulation and monitoring will reduce delivering conservation goals (Organisation for Economic Co-operation and Development (OECD), 2004). The relevance of these institutions in trading schemes is highlighted by Bovarnick et al. (2010) when stating that "the key institutional capacity building need in Latin American countries is the capacity of government to monitor and enforce third party conservation and restoration of complex habitat types to ensure that banks provide ecological equivalency to the habitat impacts the credit purchaser is attempting to mitigate".

For example, the main constraint for habitat banking in France is, as for offsets in general, the absence of standards at the national level (CDC Biodiversité, 2010). The reinforcement of standards at the national level is a condition for coherent design of offset projects in terms of duration, location and additionality. The current situation gives a competitive advantage to offset actions, which are realised with lower level of commitment for the project developer (cheaper and realised over a shorter period of time, usually 0-5 years) (eftec et al., 2010).

However, a proper instrument design and a strong capacity to attract developers and landowners to participate are also needed to ensure success. The private sector and local communities are keys for success.

As buyers, businesses create the demand that pushes forward the market; without it there is no incentive for suppliers to join the process. It is necessary to do the business case for biodiversity offsets. Voluntary schemes can a very attractive option for companies because there are significant potential benefits, such as secure license to operate and regulatory goodwill; managing risk and liability or strengthening the reputation of the company (Business and Biodiversity Offsets Programme, 2009; Crowe and ten Kate, 2010). The BBOP projects, which are voluntary offsets, are successful examples of involving companies on biodiversity conservation. However, if governments recognise that biodiversity losses are unsustainable or are compromising the ecosystems integrity, regulatory approaches can be necessary in order to achieve a greater and more consistent biodiversity outcome than will occur through voluntary approaches (Crowe and ten Kate, 2010). One example of such intervention is the Portuguese Decree-Law nº169/2001, article 8, which imposes strict conditions for the approval of cutting cork oak (*Quercus suber*) and holm oak (*Quercus rotundifolia*) and establishes specific compensation measures.

On the other hand, the suppliers of credits are mainly private landowners. Beyond this key role of providing and managing lands, landowners possess local knowledge on land characteristics, needs and opportunities, far beyond central governments or other institutions (Ring et al., 2010a). This information increases the effectiveness of the action plans and measures applied. Through the involvement of landowners this instrument can also lower the costs to achieve conservation goals.

7 The role of the instrument in a policy mix

Biodiversity offsets and banking are always linked with a regulatory framework. Habitat banking is mainly designed to address market creation and how it works, assuming that background regulation was previously established by governmental laws (Carroll et al., 2008). Typically, the laws set the rules and priorities, and then the instrument is designed to fit in its context. Regulation is the main driver for developers to offset their impacts, without regulations the demand for these markets is seriously at risk (Carroll et al., 2008). In habitat banking the demand and supply of credits rely on governments and their ability to create and enforce laws, such as federal or state endangered species acts, which are crucial to the implementation and success of such market schemes (eftec et al., 2010).

As previously referred, habitat banking is closely dependent on a regulatory framework; however, there are differences between the laws considered in each of the reported cases. Some examples of regulation applied are presented in the following.

The Department of Environment and Conservation (DEC) designed BioBanking (Australia) to support the Biodiversity Certification process under the *Threatened Species Legislation Amendment Act 2004* and to be consistent with the property vegetation planning process under the *Native Vegetation Act 2003*. DEC will use provisions in the *Threatened Species Conservation Act 1995* (including the 2004 amendments) and the *National Parks and Wildlife Act 1974* (such as conservation agreement provisions) to ensure that the scheme and management actions (offset measures) are enforceable (Department of Environment and Conservation (DEC), 2005). BioBanking is used in areas where biodiversity certification has been conferred on the planning instrument, and in other areas under specified circumstances (Department of Energy and Climate Change (DECC), 2007).

Habitat banking is a new concept in France and is currently being tested by CDC Biodiversité, a private company, which is the first conservation banking in France and launched a pilot project: “la Réserve d’actifs naturels de Cossure”. This project has no specific legal foundation and is built on established law for offsetting (CDC Biodiversité, 2010):

- The Law for the Protection of Nature (1976) introduced in France the obligation for developers to perform an Environmental Impact Assessment.
- The European Directive on Environmental Liability (Directive 2004/35/EC) integrated into national law (law 2008-757 of the 1st of August 2008 integrated in articles L160 to L165 of the Environmental Code) has clarified the definition of offset measures.
- The Forest Code (art. L311-4) plans specify rules for forest clearing.
- Rules specific to Natura 2000 sites: article 6 of the Habitat Directive transposed into articles L414-1 to L414-7 and R414-19 to R414-24 of the Environmental Code.
- Specific rules relative to the exemption from the prohibition to the destruction of protected fauna and flora species (art L411-2 Environmental Code).

In some case studies, habitat banking operates in connection with voluntary initiatives of NGO or businesses that help to reduce costs, especially those related with fieldwork, scientific research or legal requirements. For example, in Australia there are groups of lawyers that help landholders in rural areas by providing advisory and free legal assistance (e.g. preparing contracts).

Ring et al. (2010b) highlighted that the “potential trade-offs between biodiversity conservation and carbon sequestration by forests have recently attracted increased attention especially owing to the proposed financial schemes to reduce emissions from deforestation and forest degradation (REDD)”. This link between biodiversity and carbon has been already suggested in Australia, looking for a potential connection and/or merger between markets of biodiversity credits and carbon credits markets.

There are various options available to try to ensure long-term management of the offsets, including regulatory instruments, contracts (habitat banking agreements), conservation easements and funding (eftec et al., 2010). An interesting example of a mix of these instruments can be found in the Biobanking Trust Fund (Box 3) established in New South Wales. Owners of the lands included in the bank receive an annual payment out of this fund if they adequately carry out the management actions that have been set in an agreement settled between the Minister of the Environment and the owner. If not, they do not receive the payment or have to return the money paid (Department of Energy and Climate Change (DECC), 2007).

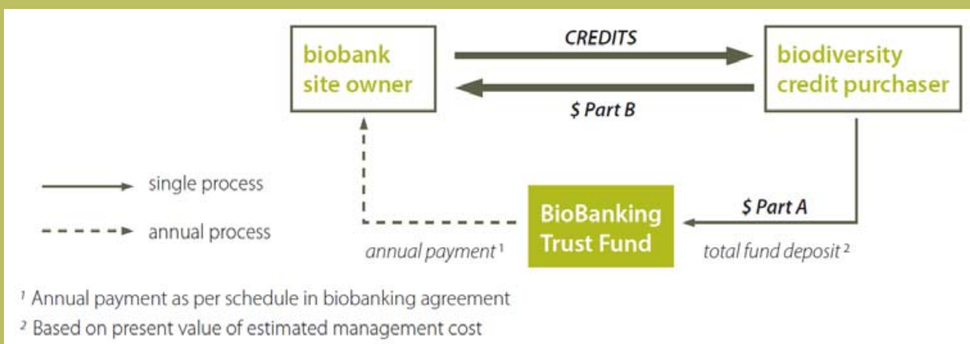
Box 3. Biodiversity Banking and Offsets Scheme (Biobanking)

It is a new option for companies to offset their impacts in a cheaper way, and also provides a new source of earnings for private landowners all over New South Wales (Australia). The main goal is to provide a market-based approach to help reduce cumulative biodiversity losses caused by population growth and development pressures around urban areas, along the coast and at major inland development sites (Department of Environment and Conservation (DEC), 2005). The biodiversity offsets are secured in-perpetuity through conservation agreements or covenants, and there are two main types of biodiversity credits – species credits and ecosystem credits.

Conservation brokers play an important role in the scheme to assist private conservation stewards to put together and market their projects. Conservation brokers might include Catchment Management Authorities (CMAs), not-for-profit organisations (e.g. the Nature Conservation Trust (NCT)), non-government organisations or conservation entrepreneurs from the private sector (Department of Energy and Climate Change (DECC), 2007).

A particular approach of this scheme is the BioBanking Trust Fund that was created to manage fund provision. It invests the funds deposited through the sale of biodiversity credits on behalf of the biobank site owners. Those funds and the investment earnings are then used to help landowners bear the costs of managing the site, over time (Department of Energy and Climate Change (DECC), 2007). Payments are made each year, after the landholder submits a report showing compliance with the agreement.

Figure 5. Credit transactions



Source: DECC - Department of Energy and Climate Change (2007)

TDR-systems are explicitly designed to operate within a planning framework (zoning). Their initial role is to compensate planning landowners in the sending zone, i.e. the area to be preserved, for restrictions imposed on potential land use. Moreover, as many authors have demonstrated – notably for air pollution control – a tradable permits-approach is unable to deal with different quality of the regulated subjects, e.g. different location of emissions or different quality of sending site-parcels without risking prohibitively high transaction costs for market participants (Atkinson and Tietenberg, 1984; Tietenberg, 1995;

Woodward, 2001). Hence, a successful permit trading scheme in land-use planning needs a strong regulatory framework to operate in (Nuissl and Schroeter-Schlaack, 2009; Woodward, 2001).

It is no surprise to find that preservation programmes with multiple goals may need multiple instruments (Lynch and Musser, 2001). Many counties within the US already combine land-use zoning, TDR and PDR to achieve the full range of goals of land-use planning. For example, even though Montgomery County has a very successful TDR-programme that already preserved a huge acreage at little cost to the county government, authorities added a PDR programme funded by tax dollars when they recognised that some of its programme goals were not being realised. Montgomery County' s PDR programme purchases easements on parcels that usually border urban areas and determines the price with a point system based on site characteristics (Lynch and Musser, 2001: 592).

It is relevant to focus the analysis on potential complementarities that might increase effectiveness or efficiency of the offset and trading mechanisms. One example is related to the uncertainties associated with offsets that entail a potential for failure. Burgin (Burgin, 2008) suggests the integration of assurance mechanisms into existing schemes (e.g. ISO14001 or the Mining Certification Evaluation Projects) in order to insure the success of the offset scheme. Additionally, the International Council on Mining & Metals (2005) suggested that "there are advantages in spreading the cost of offset development to reduce risk of failure, together with engaging external parties with interest or responsibility for offset management and design".

When compared to landscape planning, trading schemes are expected to deliver benefits in terms of cost-effectiveness and private actor engagement (Jenkins et al., 2004). However, the conditions in which it can be realised are not well understood (Ring et al., 2010a).

The introduction of a competitive bidding or auction mechanism is also a potential interesting approach. "An auction is a quasi-market institution with an interesting feature, it has a cost revealing advantage compared to PES and direct compensation payments, and can, in principle, be incorporated into a transferable development rights system" (Pascual and Perrings, 2007). This feature can help reducing information asymmetry, mainly by revealing hidden information, saving costs to the regulator, and by revealing real opportunity costs for conservation. Stoneham et al. (2003) provide data on a recent small-scale auction pilot case study for biodiversity conservation in Victoria (Australia), the BushTender.

Habitat banking is not part of the European Union policy for tackling biodiversity loss, but it can make a significant contribution to several European Union policies, e.g. Common Agricultural Policy, Habitats Directive (eftec et al., 2010). It can also contribute to tackle the cumulative fragmentation of Europe's habitats, by helping to restore, enlarge and reconnect high-nature value habitats.

Kaplowitz et al. (2008) have surveyed planning practitioners in the US about their experience with TDR. It has been asserted that communities with PDR programmes will have successful TDR programmes because on the one hand they will be more familiar with the concept of separating development rights from parcels. On the other, PDR and TDR programmes are seen as complementary; for example, using funds from one programme to leverage the other, or using one programme to target preservation in one geographical area, while using the other to target additional areas. Similarly, TDR and PDR programmes can target the same area and reinforce one another in several ways (Kaplowitz et al., 2008: 382).

In some situations it is also possible to find instruments that overlap or conflict with the habitat banking and trading schemes. To be successful, habitat banking needs to be a competitive option for developers, this means it should be a highly cost-effective solution for them to compensate their impacts. In the European Union, as in many parts of the world the existence of some perverse incentives granted by governments and other offset options can create problems to this instrument because they often correspond to cheaper solutions for developers. Regarding the European context, while there is considerable advantage in removing the perverse incentive effects of historic subsidies, few of the current laws and directives are based on a serious valuation of the social opportunity cost of biodiversity loss and fewer consider the potential effects of the introduction of new payment schemes (Pascual and Perrings, 2007).

The overlapping of multiple biodiversity conservation instruments, such as national legislation (e.g. national parks), European legislation (e.g. Natura 2000) or to international conventions (e.g. Ramsar Convention) can potentially have a negative effect on trading schemes (eftec et al., 2010). One problem is additionality, credits cannot be based on biodiversity outcomes that would have occurred anyway due to existing instruments (e.g. management obligations set up within Natura 2000 sites due to European Habitats and Birds Directives). Other issue regards the equivalence of the credits. For example, in areas with special protection status due to high biodiversity conservation values, credits should be more valuable than credits created in areas that are not classified. This crossover of different conservation legislations may induce additional constraints for project developers, leading to an increase in administrative and transaction costs (eftec et al., 2010).

Permit schemes have an obvious potential to enhance the effectiveness of land-use policy and can make a considerable contribution to the containment of land consumption, thereby facilitating biodiversity conservation. Applied within a regulatory framework of land-use planning they can fulfil three essential functions (Nuissl and Schroeter-Schlaack, 2009: 277 ff.):

Firstly, TDR can provide powerful incentives for compliance with regulatory norms and ensure that the benefits and costs of land-use controls are distributed more evenly among landowners (see also Mills, 1980). The monetary benefit for landowners to whom land development permission has been granted through zoning is (at least partially) recaptured by the requirement to hold a tradable permit. Then again, the abdication of land development – and thus the provision of the various public goods delivered by natural landscapes – is rewarded by the revenues of selling surplus development permits. This would lead to reduced government spending on monitoring and the enforcement of regulatory norms, and less welfare loss from rent-seeking activities of planning addressees.

Secondly, TDR make rigid regulative obligations set by land-use planning more flexible, maintaining or offering scope for individual compliance measures. Hence, TDR substantially augment the efficiency of land-use management previously set by planning regulation alone. This was impressively demonstrated by the huge compliance cost savings of permit trading evident in air pollution regulation in the US over the last three decades, when industry and households were forced to adapt to tighter standards for air pollutants (lead, SO_2 and NO_x) (see exemplarily Burtraw, 1996; Ellerman et al., 2000; Hahn and Hester, 1989).

Thirdly, a mixed policy that includes economic instruments is likely to enhance political acceptance of land-use control in general. Since the obligations of land-use controls must be tightened in the future in order to achieve more sustainable land-use patterns, this aspect will gain considerable significance. Again, the achievement of quite ambitious policy goals regarding air quality within a comparatively short time period in the US is a prime example here.

8 Concluding remarks

Trading schemes are a promising instrument for biodiversity conservation and there is an increasing trend towards their implementation (Wissel and Wätzold, 2010), with a higher degree of acceptance by all stakeholders, when compared to other instruments (Burgin, 2008). Governments, for example, recognise that local communities can benefit from rehabilitated sites and new income opportunities, and that it may encourage developers to participate on biodiversity conservation, overcoming business resistance to integrate conservation goals into mainstream business (ten Kate et al., 2004).

However, offsets, habitat banking and trading schemes face some theoretical and implementation problems to achieve the desired biodiversity outcomes. These are mainly related with potential market failures and strong challenges to deal with equivalence and additionality issues, which could lead to unintended economic costs and environmental consequences (eftec et al., 2010). Several scientific, management, and monitoring difficulties are also yet to be overcome (Burgin, 2010). It is possible to identify some aspects that would potentially improve the effectiveness of this instrument, namely: a) definition and use of ecological standards; b) adoption of quality indicators, instead of using only quantity indicators; and c) design and implementation of management plans.

Some features of this instrument require further research. A very relevant topic is the possibility of mitigation banks to provide multiple purposes (e.g. wetlands mitigation, endangered species mitigation and carbon sequestration). According to Bonnie (1999) this option gives higher incentives for bank owners to restore and or preserve habitats, including some that would otherwise be too expensive to acquire without government intervention. Another interesting point to discuss is on how restoration costs and time delays affect the dynamics of these schemes (Drechsler and Hartig, 2011), since habitat restoration is often costly and very time consuming.

Due to the alarming rate of biodiversity loss any effort for stabilising and reversing some biodiversity loss is important (Burgin, 2010), habitat restoration/creation does have the potential to contribute positively (Morris et al., 2006). Quoting Ring et al. (2010b), “while ecologists are still discussing whether lost areas and newly created areas are equivalent, these instruments are attempts to provide compensation”.

Permit trading is a powerful tool to realise cost-effective allocation of compliance measures, thereby minimising opportunity costs of reaching environmental goals. However, permit trading needs a clear regulatory framework to operate successfully and to obtain the anticipated efficiency gains. At local and regional level, transfer of development rights is a well established tool to contain urban sprawl and loose land development, facilitate landscape protection and promote biodiversity conservation. There is clear empirical evidence on the advantages of TDR above land-use regulation by zoning alone and the abundant literature on TDR success factors offers inspiring food for thought for policymakers.

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Tax Reliefs for Biodiversity Conservation

Frans Oosterhuis

Summary

General taxes, such as property, income and inheritance taxes, can be used in principle to provide incentives for nature protection and biodiversity conservation. The incentive can be provided by applying reduced rates or exemptions conditional on certain 'biodiversity-friendly' requirements that the taxpayer should fulfil. Examples of tax reliefs for biodiversity conservation can be found in several countries in North America and Europe, but also in some developing countries. There is only limited empirical evidence to judge the effectiveness of this instrument category. However, a general consideration pointing to restricted applicability is the fact that the basic properties of the tax scheme (taxed subjects and objects, tax rates etc.) are *not* designed with the biodiversity objective in mind. The room for a targeted biodiversity-oriented approach may often be small within the framework of these basic properties. On the other hand, tax reliefs can 'piggy-back' on existing tax mechanisms and institutions, allowing for substantial reductions in transaction costs compared to new, dedicated instruments.

1 Definition and key features

1.1 Definition

For the purpose of the present chapter, tax reliefs for biodiversity conservation can be defined as "arrangements and provisions in general tax schemes, with the explicit aim of providing positive financial incentives steering the taxpayers' behaviour in a more biodiversity-friendly direction". This definition excludes the use of tax revenues to finance or subsidise biodiversity conservation. The focus is on the taxpayers (who, as a matter of fact, might cease to be taxpayers if the incentive would consist of a full exemption). It also excludes specific environmental taxes (such as, for instance, pollution charges and taxes on the conversion of nature into urban areas).

Since tax reliefs reward the provision of ecosystem services, they might be seen as a specific type of PES.[1] However, the mechanism used to provide the incentives is quite distinct and therefore the separate treatment of tax reliefs seems justified.

1.2 Governance levels

Since taxes are by definition imposed by government, this instrument is exclusively applied by public institutions. These may, depending on the type of tax involved, act at various governmental levels (national, regional, local).

[1] *Depending on one's definition of PES. Tax reliefs would fit within PES as defined by Muradian et al. (2010) (any form of compensation for conservation behaviour), but not in the definition by Engel et al. (2008) (requiring a buyer, a seller and a contract conditional on performance). See the chapter on PES in this report.*

Table 1 shows the main types of general taxes that can be used to provide biodiversity conservation incentives, with examples and the usual level of application.

Table 1. General taxes with possible biodiversity conservation incentives and level of application

Tax type	Examples of biodiversity incentives	Common level of application
Land (use) and property taxes	Reduced rates or exemptions for forest and nature areas	Local / regional
Income and corporate taxes	Tax relief on income from forest exploitation etc.	National
	Deduction of expenses for nature management and losses on nature protection (in mixed business)	
	Preferential schemes for investment in 'green' capital goods and other assets (e.g. forest certification)	
	Tax relief on income from investments in nature and 'nature-friendly' enterprises	
Inheritance, estate, wealth, capital gains, gift and transfer taxes	Reduced rates or exemptions for forest and nature areas	National
Resource use taxes	Exemptions if resource is used sustainably (e.g. exemption from stumpage tax if trees are replanted)	National
Product taxes, excise taxes, VAT	Reduced rates for eco-labelled (certified) products	National / EU
Import and export duties	Reduced rates for eco-labelled (certified) products	National / EU

1.3 Actors involved

At the governmental level, this instrument has to be co-ordinated between the tax authorities and the public bodies responsible for biodiversity and nature protection. Generally, tax authorities tend to be in favour of taxes with a simple and stable base, and with as little as few exemptions and special arrangements as possible. From their point of view, the administrative cost of a tax should be only a small fraction of its revenues. This may conflict with the biodiversity policy makers' desire to apply tailor-made instruments that take into account the specific characteristics of particular areas and ecosystems.

As indicated above, the 'target group' at which the instrument is addressed consists of the actual tax payers. These will differ by type of tax. In any case, it is important for the policy maker to be aware of the limitations implied in the fact that actors who do not pay taxes under a particular scheme cannot be reached by biodiversity incentives under that scheme. For example, tax deductions under the corporate tax do not create any benefit for companies that do not make any profit. Likewise, VAT reductions do not affect firms using the eligible product as an input, as they can deduct VAT on inputs anyway.

1.4 Baseline

Obviously, fiscal incentives are intended to change taxpayers' behaviour. The baseline is therefore the situation (scenario) that would occur under the taxpayers' behaviour without the tax relief. This baseline behaviour may already be 'biodiversity-friendly' to some extent. This can be due to the actor's own motivations (e.g. nature protection organisations; environmentally conscious individuals; firms with a CSR policy etc.), or due to existing regulations (e.g. legislation on nature protection; restrictions on land use). In such cases, the tax incentive will be primarily aimed at encouraging and facilitating a behavioural change from 'light green' to 'dark green'.

Alternatively, the baseline might be 'gray', if the taxpayers' default behaviour has a negative impact on nature and biodiversity. Generally speaking, strong financial incentives will often be needed to turn these 'gray' practices into 'green' ones. Fiscal reliefs will only in exceptional cases be capable of providing such strong signals.

1.5 Range of application

The scope for tax reliefs is in principle determined by the scope of the various taxes that are levied from actors whose behaviour affects biodiversity. Since taxes are ubiquitous, tax incentives can be widely applied in principle. The most obvious application areas are those where there is a direct relationship between the taxed object and the specific biodiversity values to be protected. For example, land and property taxes can be differentiated according to the type of land use, with low or zero rates for 'biodiversity friendly' types of land use. However, it is also possible to use more general taxes (such as income taxes or VAT[2]) as the payment vehicle. Basically, tax reductions can be given for any kind of socially desirable behaviour. As such, tax reliefs are essentially just a particular way of subsidising activities with 'public good' features or positive external effects.

2 Literature review of the instrument's performance

2.1 Environmental effectiveness

The effectiveness of tax reliefs can be measured in various ways. Usually, it will not be possible in practice to measure the 'ultimate' effectiveness in terms of, for instance, the area of a specific habitat or the number of a certain species preserved (compared to the situation without the scheme). A more feasible way to assess effectiveness is to determine the extent to which the scheme has induced the envisaged behavioural changes (such as taking specific nature-friendly measures or refraining from certain land-use changes). This can be expressed, for instance, in terms of the percentage of eligible landowners or area of eligible land actually enrolled under the tax relief scheme.

2.1.1 Empirical evidence

Until now, there has been no systematic survey of global experiences with this type of instrument. The evidence on the effectiveness of tax reliefs for biodiversity conservation is limited to a small number of case studies. Below, some examples are presented from various countries.

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[2]*For example, until its accession to the EU the Czech Republic applied a reduced VAT rate to several kinds of 'green' goods, including recycled paper (with a potential positive impact on forest conservation).*

Canada

In 1996, the Income Tax Act was amended to exempt from capital gains tax all donations of ecologically sensitive lands made in perpetuity to all levels of government and charities. Presently, the Income Tax Act provides favourable tax treatment for the disposition of ecologically sensitive land (including a covenant, an easement, or a servitude) to Canada, a province or territory, a Canadian municipality or a registered charity that the Minister of the Environment has designated. This treatment includes a reduction in capital gains realised on the disposition of ecologically sensitive land and the provision of a tax credit or a deduction to donors, up to 100% of their net income (source: www.cra-arc.gc.ca).

From the outset, this 'Ecological Gifts Programme' has been considered a successful example of the integration of fiscal and environmental policies to encourage conservation of biodiversity on private and corporate-owned lands (OECD, 1999). By 2010, over 830 ecological gifts valued at over US$ 535 million had been donated across Canada, protecting over 136,000 hectares of wildlife habitat. More than one-third of these ecological gifts contain areas designated as being of national or provincial significance, and many are home to some of Canada's species at risk (source: www.ec.gc.ca).

United States

Fiscal incentives for biodiversity conservation exist both at federal and state level (Defenders of Wildlife, 2006). Federal income tax incentives include deductions for donating conservation easements (legally binding restrictions placed on a piece of property to protect its resources by prohibiting specified types of development from taking place). Federal estate taxes (paid on inherited property) are reduced for land under conservation easement. By 2005, land trusts held conservation easements on over 2.5 million hectares and government agencies and national non-profit organisations also had sizable holdings under easement (TEEB, 2009: section 5.4). There are also federal tax reliefs for incurring conservation expenditures, and for revenue derived on lands that are managed to support natural habitat.

Most of the states also provide some form of state tax benefit for citizens that maintain wildlife habitat, but not all of them are very effective. For example, the Indiana Department of Natural Resources (IDNR) Division of Forestry administers a programme known as the Classified Forest Program (CFP) adopted by the state, with the goal to "keep Indiana's private forests intact." Classification of a forest results in a reduction of assessed value for state property taxes. The incentive of the programme is relatively weak, as it provides property tax relief in a state where property taxes are very low, particularly on forestland. In 1998, there were 8300 properties enrolled in the programme, covering 410,000 acres owned by 6300 landowners, which is 10% of private forested acres and less than 5% of the estimated 150,000 forest owners (York et al., 2006).

LeMense Huff (2004) analysed tax incentive schemes for conservation in two states (Virginia and Oregon) and concluded that it may be more beneficial if fewer lands were carefully selected for conservation, and those lands fully evaluated, inventoried, monitored and thereby protected. According to the author, this approach would possibly decrease overall monitoring costs and increase the value to taxpayers.

Bolivia

Bolivia has an active policy to promote (FSC) certification in forestry. Certified concession-holders enjoy tax benefits of 14–28%, which roughly offset direct certification costs. This is seen as one of the factors contributing to the much larger uptake of forest certification in Bolivia than in Ecuador, where such incentives are lacking (Ebeling et al., 2009).

South Africa

In the past, the National Parks Act prescribed that no rates or taxes of any kind should be levied on any land or any building that is situated within a national park. This provided a significant incentive for individuals and communities to contract their land to national parks. However, the National Parks Act was repealed by the Protected Areas Amendment Act, which did not preserve the above exemption (Paterson, 2005).

South Africa's Income Tax Act provides that expenditure incurred in the eradication of alien and/or invasive vegetation can be deducted for income tax purposes. This is a potentially valuable tool for the government to share the costs of clearing this vegetation with private landowners. However, its effectiveness is limited by restricted eligibility in terms of landowners and activities (Paterson, 2005).

The Netherlands

Green Funds
Since 1995, the 'Regeling Groenprojecten' (Green Funds Scheme) offers a fiscal incentive to invest in green projects, including (among others) projects in the areas of nature, forestry and organic agriculture. In order to be eligible, projects need a 'green certificate', issued by the Ministry of Environment. Presently, all major banks have dedicated 'Green Funds', investing in eligible green projects. Money invested or saved in such 'Green Funds' is exempt from income tax. This allows the bank to pay a lower dividend (or interest rate) on such investments (savings), and thus to charge a lower interest rate on the money lent to the project initiator, or to accept a lower level of profitability.

By the end of 2009, more than 6000 projects had received a 'green certificate'. Among these were almost 800 projects in the categories 'nature, forests and landscape', and 1600 projects in the category 'organic agriculture'. The total amount of investments in these (biodiversity related) projects was € 1.9 billion (Agentschap NL, 2010).

The effectiveness of the Green Funds Scheme was assessed in 2002 (KPMG and CE, 2002). The impact on biodiversity could not be quantified. The study noted, however, that on organic farmland certain species are found in significantly higher numbers than on conventional farmland. On the other hand, other projects eligible under the Green Funds Scheme (such as wind turbines) can have a (small) negative impact on biodiversity (additional bird deaths).

Donations
As in many other countries, donations by private persons to (officially recognised) organisations pursuing a public interest (including nature protection organisations) can, under certain conditions, be deducted from taxable income. In 2009 this 'gifts deduction' scheme was evaluated (Ministerie van Financiën, 2009). The analysis found that the impact of the fiscal incentive scheme on the number of taxpayers donating, or on the size of their donations, is not statistically significant.

France

In France, a number of tax reliefs have been introduced recently that may be beneficial for biodiversity. There are reductions and exemptions from the property tax (taxe foncière) for undeveloped property in Natura 2000 sites and wetland areas[3] and for land that is used for organic agriculture[4]. The tax relief is for

[3] *Articles 137 et 146 de la loi N° 2005-157 du 23 février 2005.*

a period of 5 years and is conditional on a commitment by the property owner to comply with certain management practices. The size of the financial incentive seems to be limited though: in one example it was calculated at just € 5 for 10 hectares of peatland.[5]

2.1.2 General considerations on effectiveness

Apart from the empirical evidence, a number of constraints can be identified that limit the *potential* effectiveness of tax incentives:

- Obviously, a tax reduction can only be applied effectively if a tax exists in the first place, and applies (in principle) to the land that holds the conservation value (or its owner). Some states in the USA, for instance, have no income tax and therefore also not the option to reduce it for conservation purposes. Likewise, in several countries, land used for agriculture, horticulture and forestry is not subject to land or property taxes. Especially in developing countries, the fiscal system leans heavily on indirect taxes (trade taxes and VAT), which provide little opportunities for biodiversity oriented tax reliefs. In many countries a significant part of protected areas is owned by public entities that do not pay income or corporate taxes. Furthermore, even if the landowner is liable to pay taxes in principle, his net income or profit may be zero or even negative (a situation not uncommon in nature management and forestry), which means the tax reduction will have no impact.[6] Similarly, reductions in taxes that are based on the value of the land (such as property taxes) will often have limited impact, since the value of land under use restrictions will already be low to begin with.
- Likewise, tax reliefs will only be effective to the extent that tax payments are actually enforced. In countries where tax evasion and informal economic activities are pervasive, the 'effective' tax rate may often be zero already in the baseline situation, implying that the introduction of tax reliefs does not offer additional incentives.
- Effectiveness will (*ceteris paribus*) be higher if tax rates are higher. Thus, although in the United States a relatively broad set of fiscal incentives is applied, their effectiveness is limited by the low tax rates in that country (compared to Europe). Income tax deductions provide the strongest incentives to taxpayers with high income (except in countries with a 'flat tax' regime).
- Tax reliefs do not (or only to a limited extent) allow for a targeted approach as far as the amount of conservation offered and the conditions for eligibility are concerned. A specific tax has a limited number of tax rates, and applying a reduced rate (or an exemption) would only coincidentally provide exactly the financial incentive needed to make the landowner or other actor change his behaviour. Moreover, tax law principles require the conditions and criteria for tax relief to be formulated in general terms that do not discriminate between individual taxpayers. It is therefore a rather random way to select properties for conservation, leaving little room for discretionary action in terms of selecting particular sites that would qualify (e.g. because of their function in achieving ecosystem integrity or as an ecological 'hotspot' or corridor). As a result, the configuration of lands conserved in response to tax incentives can be sporadic and unpredictable, which may diminish the value of the lands conserved from an ecological standpoint (Boyd and Simpson, 1999; Clough, 2000; LeMense Huff, 2004).

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[4] *Loi N° 2008-1425 du 27 décembre 2008.*

[5] *Réseau SAGNE Midi-Pyrénées-Tarn, Lettre de liaison n° 14, Septembre 2008.*

[6] *However, a tax relief scheme may contain a provision that a taxpayer who is unable to use an otherwise allowable tax credit may transfer the unused credit for use by another taxpayer. LeMense Huff (2004: 145) mentions the example of income tax credits in Virginia.*

- Just like other financial incentives, tax reliefs are based on the assumption that private decisions with consequences for biodiversity are taken primarily on the basis of financial cost-benefit considerations. To the extent that other factors (e.g. tradition, culture) dominate, the effectiveness of tax reliefs will be limited.

2.2 Cost-effectiveness and efficiency

Generally speaking, cost effectiveness is a main advantage of fiscal incentives over other instruments, such as specifically designed subsidy schemes. Fiscal incentives 'piggy-back' on existing tax mechanisms and thus a substantial part of the transaction costs associated with other instruments can be avoided.

On the other hand, tax relief is not a zero cost option. In addition to the tax revenues foregone (which may have to be compensated for by increasing other taxes that create economic distortions), there are additional costs of monitoring and enforcing compliance with the conditions for tax relief. Saving on the latter costs will often mean a higher risk of abuse and fraud, which imply social costs as well.

Another limitation on cost-effectiveness is the phenomenon of 'free riding': the tax benefits are available to all actors who meet the conditions, including those who would have acted in a 'biodiversity-friendly' manner anyway, even without the financial incentive.

The cost-effectiveness and efficiency of a tax relief scheme will also be reduced if its design contains certain 'perverse incentive' elements. For example, LeMense Huff (2004) notes that preferential assessment under a property tax (in which land is assessed on the basis of its current use rather than its most profitable potential use) may create such a perverse incentive. Once the landowner changes the use so that the property no longer qualifies for preferential tax treatment, the property tax benefit disappears, but the landowner pays no penalty. This may imply that taxpayers reduce costs for speculators without any lasting conservation benefit.

2.3 Social impacts

Tax reductions and exemptions generally convey larger benefits to the rich than to the poor. This is in particular the case with income tax breaks, since most countries apply a progressive rate structure (rates increasing with income level).

From a social justice point of view, tax reliefs (just like PES and other types of subsidy) could be criticised for the fact that they convey a financial reward to landowners for something that they should (or would[7]) do anyway. Obviously, this depends on the (normative) 'baseline' (see section 1.4): what level of biodiversity protection can be considered as a basic duty for the landowner? Any public money for protective measures up to that level could be said to violate the 'polluter pays principle'.

2.4 Institutional context and requirements

As noted above, tax incentives can be built into an existing institutional framework. The basic elements of a tax system (legislation, authorities, procedures, capacity, expertise) are already there when introducing a conservation incentive into an existing tax scheme. Obviously, these will need to be supplemented by specific elements related to the new provisions.

[7] *I.e. free riders; see above.*

The governance level at which tax reliefs for biodiversity conservation are applied deserves due consideration. For instance, if local taxes (e.g. land taxes) are reduced, the local government loses income. This may be seen as unfair, given the fact that conserving biodiversity has a national or even global value. Reluctance among local authorities to foot this bill may imply the need for the central authorities to look for ways of compensating them, e.g. by changing the existing rules for the allocation of public funds to lower levels of governance (fiscal transfers; see Ring et al., this report). In France, for instance, the state compensates local authorities for the loss of tax revenue due to property tax exemptions for wetlands and Natura 2000 sites.

3 The role of the instrument in a policy mix

Doremus (2003) argues that a portfolio approach, creatively and flexibly combining many of the various available biodiversity conservation policy strategies, holds the most promise for addressing a wide variety of conservation problems in different geographic and social contexts. Within this portfolio, financial incentives have a role to play alongside other instruments such as regulation, education and market creation.

Among the financial incentives, tax relief can play a role as an instrument that makes certain types of biodiversity enhancing actions financially more attractive. It does so in a general way: for fine-tuning to achieve specific conservation targets the use of other incentives is more appropriate.

Tax reliefs from land and property taxes are often limited to land that already has some kind of official status as protected area. In such situations, the fiscal benefit does not act as an incentive, but rather as a compensation for the loss of economic value associated with the use restrictions imposed on the landowner.[8] Clearly, the regulatory and the economic instrument have complementary functions here instead of being alternatives.

4 Concluding remarks

Tax reliefs can be an attractive set of tools in the biodiversity conservation toolbox, primarily because they can to a large extent use the existing fiscal infrastructure for administration, assessment, payment, monitoring and enforcement. The transaction costs of this instrument are therefore relatively low, but the benefits (in terms of effective biodiversity conservation) may also be low and uncertain. In particular the latter aspect (uncertainty) will be an important consideration when deciding on the use of fiscal incentives.

Generally speaking, the following ‘situation characteristics’ could be favourable conditions for a potentially cost-effective application of tax relief:

- The geographical location of the land benefiting from the tax relief is relatively unimportant; in other words: the conservation efforts promoted by the incentive are equally valuable, regardless of where they take place.
- If location does matter, the conditions for tax relief can be specified in a way that makes only the relevant sites eligible (but this will generally be difficult, as indicated above).

[8] *Strictly speaking, this type of tax relief does not meet our definition given in section 1.1.*

- Complete coverage of a certain area is not necessary; in other words: it does not matter if some landowners in the area do not participate in the conservation.
- The financial incentive provided by the tax relief is in the same order of magnitude as the additional cost (or revenue foregone) of the conservation actions aimed at.

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Ecological Fiscal Transfers

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Summary

Intergovernmental fiscal transfers redistribute public revenue from national and state governments to local governments. Comparatively new is the rational to use fiscal transfers for conservation policies. Although recommended in a number of countries for introduction, to date, only Brazil and, more recently, Portugal have implemented fiscal transfers for biodiversity conservation. In this chapter, we analyse this innovative policy instrument, building mainly on a review of existing experiences in a number of Brazilian states and Portugal. We develop definitions in terms of the effectiveness, cost-effectiveness, and distributional impacts of ecological fiscal transfers and specify their role in a wider policy mix for conservation.

1 What are ecological fiscal transfers?

1.1 Definition and key features

Under fiscal transfer schemes, public revenue is redistributed through transfers from national and subnational governments to local governments. Intergovernmental fiscal transfers help lower-tier governments cover their expenditure in providing public goods and services. In developing and transition economies, about 60 % of subnational expenditure is financed by these transfers, in non-Nordic Europe and Nordic OECD countries, they account for 46 % and 29 %, respectively (Shah, 2007). Another purpose of such schemes is to compensate decentralised governments for expenditure incurred in providing so-called spillover benefits to areas beyond their boundaries (Olson, 1969). The bulk of fiscal transfers is allocated in the form of lump-sum or general purpose (unconditional) transfers. The recipient government is free to decide upon their use and thus, local autonomy is preserved. In many countries, fiscal capacity (own source public revenue) and fiscal need (based on specified indicators such as population or area) of a subnational government determine the amount of transfers received, introducing a distributive element in the form of “fiscal equalisation”. In addition, there are specific-purpose (earmarked or conditional) transfers that are only allocated for the provision of specific public goods and services.

Ecological fiscal transfers are allocated on the basis of ecological or conservation-based indicators, such as protected areas (see Box 1 for an overview of rationales for introducing ecological fiscal transfers). They may be allocated in the form of lump-sum or specific-purpose transfers. In addition, ecological fiscal transfers represent any earmarked transfers for ecological or conservation purposes. These latter earmarked transfers have been used more commonly in intergovernmental fiscal relations in many countries, especially for end-of-the-pipe and infrastructure-related ecological public functions such as sewage and waste disposal (Ring, 2002). Comparatively new is the rationale for ecological fiscal transfers in biodiversity conservation that is the focus of this review. Decisions about where conservation areas are to be sited are frequently taken at higher levels of government, even though the costs of losing those areas for other social and income-generating developments are borne by the local governments and communities. Fiscal transfers are therefore seen as an innovative instrument to provide incentives for local governments to support and maintain the quality of water and nature conservation areas within their territories, but which can also provide wider ecological benefits beyond municipal boundaries (Ring, 2008a; ten Brink et al., 2011).

Last but not least it is important to state that ecological fiscal transfers are but one (economic) instrument in the mix of relevant policy instruments for biodiversity conservation. Box 1 summarises the different possible rationales for introducing ecological fiscal transfers. In this chapter's review, we exclusively focus on ecological fiscal transfers addressing public actors, i.e. local and state governments. Specific purpose transfers may also take on the form of Payments for Environmental Services (PES) to non-governmental actors such as land users (see 3.2 in Box 1). These governmental support programmes are not covered here, but are part of the "payments for environmental services" chapter (see Porras et al., this report).

Box 1. Different possible rationales for ecological fiscal transfers

1. Compensation of expenses/supply costs for ecological public goods and services
2. Compensation of opportunity costs
    - 2.1 Loss of land-use revenue on municipal property
    - 2.2 Loss of tax revenues from private landowners prevented from doing business
3. Payments for external benefits
    - 3.1 to local governments for providing spillover benefits beyond their boundaries
    - 3.2 to non-municipal stakeholders within municipal boundaries
4. Fiscal equalisation / distributive fairness
    - 4.1 Vertical equalisation between higher and lower levels of government
    - 4.2 Horizontal equalisation between jurisdictions at the same level of government

Financial transfers at the international level are usually discussed under the term International Payments for Ecosystem Services (IPES). They provide a comparable mechanism to account for costs and spillover benefits of conservation at the international scale (Farley et al., 2010). Whereas fiscal transfer schemes within a nation state are based on financial constitutions and are highly regulated by laws, IPES so far are based on voluntary action by donating governments. However, such schemes promise to play a role in the context of the recently discussed REDD and REDD-plus schemes on Reducing Emissions from Deforestation and forest Degradation (Ring et al., 2010; cf. Chacón-Cascante et al., this report).

1.2 Relevant actors

Whereas Payments for Environmental Services schemes are mostly used to compensate the conservation costs born by land users, i.e. private actors, ecological fiscal transfers address public actors. In centralised states, such as Portugal, ecological fiscal transfers are allocated from the national level to municipal governments, as defined in the new Local Finances Law as of January 2007 (Santos et al., 2010). In federal systems, such as Brazil and Germany, there is an intermediate state level. A fiscal transfer scheme exists between the national level and the states (depending on the country also called regions, provinces or *Länder*), and again between each state level and its local governments. Within certain limits set by the federal constitution, federalism in these countries allows the state governments to define their own fiscal transfer schemes. So far, no federally organised country has implemented ecological fiscal transfers based on conservation indicators between the national and state level, although there have been proposals suggesting such schemes (Silva, 2000; Czybulka and Luttmann, 2005). Brazil, however, is the first country that has introduced ecological fiscal transfers (ICMS Ecológico) in a number of states to compensate municipalities for land-use restrictions imposed by protected areas (Grieg-Gran, 2000; Loureiro, 2002; May et al., 2002; Ring, 2008a).

Fiscal transfer schemes are part of a country's or state's constitution and subsequent implementing laws and regulations. For any government budget heavily relies on the rules as set out in relevant legal and institutional frameworks, the design and amendments of such laws are highly politicised processes, involving concerned governments and their associations (for example, association of localities and districts). Lead responsibility in terms of expert knowledge, design and implementation lies in the realm of financial ministries. In the case of ecological fiscal transfers, policy design, implementation and monitoring may be supported by environmental ministries and conservation or forest authorities.

1.3 Baseline

Existing ecological fiscal transfers in Brazilian states and Portugal use officially designated protected area and/or areas designated as water catchments as indicators to allocate lump-sum transfers to local governments. So the baseline for this instrument type may be interpreted as the amount of designated protected area when the instrument is introduced, though the lump sum payment is adjusted each year in recognition of additional protected areas that come into being. In most schemes, just the protected area coverage as a quantitative indicator is used. In Brazil, the different categories of protected areas are further multiplied by a conservation factor or weight, reflecting the varying land-use restrictions associated with, for example, strictly protected or sustainable use areas (Grieg-Gran, 2000; May et al., 2002; João, 2004; Ring, 2008a). In Portugal, transfers per hectare protected area are higher, if protected area coverage in relation to municipal area is beyond 70 % (Santos et al., 2010). Some Brazilian states, such as Paraná and Minas Gerais, have additionally introduced a quality indicator into relevant legislation. Thus far, only Paraná has rigorously implemented the quality indicator that may range from 0 to 1. So if a protected area is badly managed or just a "paper park", the transfers associated with it may in theory even drop to zero. Other states, such as Mato Grosso, have not explicitly introduced a quality indicator. However, ICMS Ecológico legislation in Mato Grosso states that if monitoring by the State Environmental Foundation "provides evidence of degradation for characteristics which justify the protection of a conservation unit or indigenous territory, the conservation factor must be reduced by 50 %" (Skiba, 2010: 20). In this way, protected area quality is also accounted for, although the method used is less detailed compared to the explicit quality factor in Paraná.

Both in Brazil and Portugal, ecological fiscal transfers have been introduced to compensate municipalities for the land-use restrictions associated with protected areas. Depending on the protected areas' category, economic development is more or less restricted leading to opportunity costs. There are few studies that try to account for the municipal benefits forgone in terms of lower tax revenues from restricted economic activities (Azzoni and Isai, 1994). In practice, it is very difficult to exactly determine the total costs (management, opportunity and transaction costs) and benefits at different spatial levels associated with protected areas.

After all, recognising the positive spatial externalities associated with protected areas provides a positive incentive for municipalities to acknowledge and value the natural capital within municipal boundaries that is otherwise mostly perceived as an obstacle to development (Stoll-Kleemann, 2001). No recognition of positive externalities clearly leads to an underprovision of the relevant public goods and services, in this case nature conservation (Bergmann, 1999). Thus, the increase in protected areas in a number of Brazilian states after the introduction of ecological fiscal transfers (see below) may on average be seen as the equivalent amount of additional nature protection in response to the financial incentive offered to municipalities. In this context, it is relevant to have a look on the type of protected areas that grow in numbers and ideally also on the quality associated. It is also important to check whether the baseline area of protected areas was static or in an upward trend at the time of the instrument's introduction in the relevant state.

1.4 Range of application of ecological fiscal transfers

Fiscal transfers for environmental purposes have been used for many years. This is especially the case for infrastructure-related public services such as drinking water provision or sewage and waste disposal, which are traditional areas of local public service provision to citizens (Ring, 2002; Ring, 2008b). However, major differences still exist in the provision of these public services between developed and developing or transition economies that involve environmental infrastructure capital investments. Whereas in a country like Germany, drinking water provision or sewage and waste disposal have long been addressed as basic services by municipalities, in a country like Brazil, much remains to be done in many places to provide citizens with high quality drinking water and healthy sanitary systems. Therefore, as part of the ICMS Ecológico, a number of Brazilian states have also introduced indicators related to waste disposal or healthy sanitation systems.

To date, only Brazil and, more recently, Portugal have implemented fiscal transfers for biodiversity conservation. In a number of other countries, such transfers have so far been only proposed, and potential consequences partly modelled (for Germany: Ewers et al., 1997; Ring, 2002, 2008b; Perner and Thöne, 2005; for Switzerland: Köllner et al., 2002; for Indonesia: Mumbunan et al., 2010). In some countries there are also proposals to use ecological fiscal transfers for forest conservation, such as in India (Kumar and Managi, 2009) or Indonesia (Irawan and Tacconi, 2009; Ring et al., 2010).

So far, the achievement of ecological or conservation-related objectives remains a rather minor part of intergovernmental fiscal transfer schemes. It is in fact a timely task of environmental policy integration to increasingly mainstream ecological next to social and economic public functions in intergovernmental transfer schemes. As the introduction to this section has shown, fiscal transfers are core to fiscal and social policies and used to equalise public revenues across rich and poor jurisdictions according to their fiscal capacities and needs. This opens an alley for fiscal transfers to combine conservation policies with poverty alleviation objectives (OECD, 2005), an important characteristic for designing policies in developing and transition countries.

2 Policy analysis of existing schemes: The ICMS Ecológico in Brazil

2.1 Introduction

Regarding the integration of conservation-related indicators into intergovernmental fiscal transfers, most practical experience exists in the various Brazilian states having introduced the ICMS Ecológico (ICMS-E). To date 16 out of 26 Brazilian states have introduced the ICMS Ecológico in their states' constitutions, while 13 have actually implemented ecological fiscal transfers on this basis through appropriate enabling legislation (Table 1). Therefore, policy analysis of ecological fiscal transfers with respect to their environmental effectiveness, cost-effectiveness, social impacts and institutional context will be predominantly reviewed based on available literature and information on the various schemes in Brazilian states. There will be a separate subsection on the recently introduced new Portuguese Local Finances Law and another on ecological fiscal transfer schemes that have only been suggested in the literature.

Table 1. Brazilian states with ICMS Ecológico approved, in an operational phase or undergoing regulatory authorisation, in order of year created

State	Year	Environmental criteria	
		Biodiversity conservation (%)	Other criteria (%)
Paraná	1991	2.5	2.5
São Paulo	1993	0.5	0.0
Minas Gerais	1995	0.5	0.5
Rondônia	1996	5.0	-
Amapá	1996	1.4	-
Rio Grande do Sul	1998	7.0 ([1])	-
Mato Grosso	2001	5.0	2.0
Mato Grosso do Sul	2001	5.0	-
Pernambuco	2001	1.0	5.0
Tocantins	2002	3.5	9.5
Acre ([2])	2004	20	-
Rio de Janeiro	2007	1.125	1.375
Goiás ([2])	2007	5.0	-
Ceará	2007	-	2.0
Piauí	2008	-	5.0
Pará ([2])		Undergoing definition	Undergoing definition

Sources: State legislations.
Notes: (1) In Rio Grande do Sul, the criterion is triple the area protected; (2) In Goiás, Pará and Acre, constitutional amendments have provided for creation of the mechanism, but there is as yet no enabling legislation.

The ICMS Ecológico was created in 1991 in Paraná, as a measure to compensate municipalities which faced opportunity costs from revenue loss due to watershed protection for water supply to the larger Curitiba metropolitan area. Rather than make such compensation restricted to this area, legislators determined to extend it to the entire state, and to include an equal share for other protected areas. Such areas include public and private areas protected under the national system of protected areas (SNUC, which came into effect in 2000) as well as locally relevant common property forest areas known as *faxinais.* The allocation formula adopted contains both quantitative and qualitative variables. The former refers to a combination of proportional area designated for protection in relation to each municipality's physical territory. Secondly, distinct categories of protection are afforded different compensation weights under the Paraná system – so-called conservation factors –, reflecting the degree to which other uses are excluded. Thus, parks, reserves and ecological stations are afforded the highest weight (1.0), whereas environmental protection areas (APAs) and private nature reserves (RPPNs) which offer some options for direct use by local populations are given a much lower weight (0.1)[1].

The qualitative aspect of the Paraná allocation scheme is unique, referring to aspects which are judged to improve the relative degree of conservation integrity of those areas protected within a given municipality. These are visited periodically by state personnel to appraise the extent to which buffer zones and the overall municipality exhibit uses compatible with the level of protection afforded, as well as other variables, including the existence of endemic species of flora and fauna, and inputs provided by the local government to ensure maintenance and improvement in the prior levels of management and protection.[2]

[1] Although RPPN's are much more restrictive than APAs, there is a tendency to downgrade them since their protection is usually adjacent to direct uses by the same landowners.
[2] As an orientation for the construction of the group of qualitative variables they include such aspects as biological and physical quality and especially those associated with planning, implementation, maintenance and management

None of the other states which have implemented ecological ICMS legislation have developed as complete a weighting scheme based on quality though a number of states have included qualitative adjustment as a possible tool for conservation improvements. This is due to the additional costs involved in introducing a quality indicator. It requires regular monitoring of conservation areas, training of field staff and implies that field staff is trusted to make impassionate non-subjective qualitative judgements.

2.2 Effectiveness of biodiversity conservation

Thus far, environmental effectiveness of ecological fiscal transfers has not been explicitly addressed in the literature. Existing schemes have been primarily introduced to compensate for the land-use restrictions associated with existing protected areas. In this way, there is no new environmental objective to be achieved with the policy instrument that may be evaluated using criteria of traditional policy analysis. However, as the Brazilian example has shown, ecological fiscal transfers may develop as an incentive and municipalities react on the newly introduced ecological indicator. For ecological fiscal transfers, environmental effectiveness may thus be interpreted as the increase of the newly introduced ecological indicator after the implementation of the new scheme. In terms of biodiversity conservation this relates to the increase in quantity and quality of the various categories of protected areas in the relevant jurisdiction having implemented the ecological fiscal transfer scheme.

Crucial factors for success in terms of effectiveness for biodiversity conservation certainly relate to the characteristics of the ecological indicators used. All Brazilian states with ICMS Ecológico have introduced the quantity of conservation units in relation to municipal area as the basis for transfer distribution. Regarding the various categories of protected areas, weightings have been introduced to favour strictly protected areas with high land-use restriction over sustainable use areas with low land-use restrictions involved. Thus, higher transfers are issued for parks and reserves compared to lower transfers for environmental protection areas. Table 2 provides an overview on conservation factors used for different management categories of protected areas. Due to fiscal and environmental federalism in Brazil these factors differ for different Brazilian states. As indicated before, only Paraná has also introduced and implemented detailed quality-related criteria that contribute to conservation effectiveness and provide a strong incentive for quality management of protected areas.

Although the ICMS Ecológico has originally been introduced as a compensation for land-use restrictions, it developed at the same time as an incentive to create new protected areas (May et al., 2002). This can be demonstrated for those states possessing the ICMS-E for a number of years. Ribeiro (2008) presents increase in protected area coverage for municipal, state and federal conservation units since the introduction of the ICMS Ecológico for a number of Brazilian states.[3] More detailed empirical analyses exist for Paraná and Minas Gerais, two states having introduced the ICMS Ecológico comparatively early (Grieg-Gran, 2000; Loureiro, 2002; May et al. 2002; Veiga Neto, 2000). For example, recent numbers for Paraná indicate that in total, protected areas have increased by 164.5 % since the introduction of the ICMS Ecológico (Table 3). This increase has mostly taken place within the first 10 years after the instrument had been introduced (cf. increase presented by May et al., 2002), indicating a certain saturation effect.

of the protected area, along with meeting requirements for infrastructure, equipment (audiovisual, support), personnel and training, research underway, legitimacy of the protected area as perceived by the local community and extend to a supplemental analysis of municipal government in the areas of housing and urban planning, agriculture, health and sanitation, as well as support to agriculture and local communities.

[3] *Bachelor thesis, Universidade Federal Rural do Rio de Janeiro, see also TNC (2010).*

Table 2. ICMS Ecológico: Conservation factors for different management categories of protected areas in different Brazilian states

Management categories	Conservation Factors						
	MG	**MT**	**MS**	**PR**	**PE**	**SP****	**TO**
1. Ecological Station	1.0	1.0	1.0	0.8/1.0*	1.0	1.0	1.0
2. Biological Reserve	1.0	1.0	1.0	0.8/1.0*	1.0	1.0	1.0
3. Parks (National, State, Municipal)	1.0	0.7	0.9	0.7/0.9*	0.9	0.8	0.9
4. Natural Monument	1.0	0.8	0.9		0.7		0.8
5. Wildlife Refuge	1.0	0.8	0.9		0.75		0.8
6. Private Natural Heritage Reserve (RPPN)	1.0	0.2	0.7	0.68	0.8		0.6
7. Forest (National, State, Municipal)	0.3	0.5	0.6	0.64	0.6	0.2	0.5
8. Environmental Protection Area (APA) with management plan	0.1	0.2	0.05		0.1-0.7	0.1	0.1
9. Environmental Protection Area (APA) with no management plan	0.025	0.2	0.05	0.08	0.05	0.1	0.1
10. Area of Relevant Ecological Interest (ARIE)	0.3	0.3	0.08	0.66	0.45		0.4
11. Wildlife Reserve	0.3	0.4	0.6		0.6		0.4
12. Sustainable Development Reserve	0.5	0.5	0.05		0.4		0.2
13. Extractivist Reserve	0.5	0.5	0.4		0.5		0.45
14. Indigenous Reserve	0.5			0.45			0.5
15. Area of Relevant Touristic Interest (ARIT)				0.08			
16. Tourism Destination							
17. Buffer Zone							
18. Wildlife Zone in APA	0.5					0.5	
19. Restricted Use Areas						0.1	
20. Parkway		0.3					
21. Indigenous Territory	0.5	0.7	0.45				
22. Special Protected Area		0.5					
23. Scenic Rivers			0.24				
24. Scenic Roadways			0.08				
25. Natural Resources Reserve			0.8				
26. Ecological Reserve					0.3		
27. Private Land Restoration Area (RPRA)	0.1						
28. Faxinais				0.45			

* *Higher values for locally protected areas, lower values for state and federally protected areas.*
** *Legislation in São Paulo only acknowledges state conservation units for ICMS Ecológico transfers.*

According to the national system of protected areas (SNUC), management categories 1–5 belong to conservation units with so-called integrated protection, involving high land-use restrictions. Management categories 6–13 belong to sustainable use areas, involving lower land-use restrictions. The other management categories included in the table are not classified in federal legislation.

Sources: MG/Minas Gerais: Minas Gerais (2009); MT/Mato Grosso: Mato Grosso (2000); MS/Mato Grosso do Sul: Mato Grosso do Sul (2001); PE/Pernambuco: Pernambuco (2003); PR/Paraná: Loureiro (2002: 168); SP/São Paulo: São Paulo (1993); and TO/Tocantins: Tocantins (2002).

In Minas Gerais, where the ICMS Ecológico was introduced in 1995, protected areas increased by 62 % by the year 2000 (Veiga Neto, 2000; May et al., 2002; Ring, 2008a). However, this increase in protected areas was in part due to the efforts by local governments to register already existing protected areas that had not been regulated previously by the state within the first year after the introduction of the scheme (Bernardes, 1999).

Table 3. Growth in protected areas up to 1991 and from 1992 to 2009, Paraná

Protected areas	Prior to 1991 (ha)	Up to August 2009 (ha)	Increase (%)
Federal conservation units	584,622.98	714,913.10	22.3
State conservation units	118,163.59	970,639.05	721.4
Municipal conservation units	8,485.50	231,072.02	11,338.8
Indigenous areas	81,500.74	83,245.44	2.1
RPPN	0	42,012.09	0
Faxinais (traditional community)	0	17,014.56	0
Permanent Protection Areas – APP	0	17,107.69	0
Legal Reserves – RL	0	16,637.73	0
Special Sites – SE	0	1,101.62	0
Other connective forests – OFC	0	3,245.62	0
Total	792,772.81	2,096,988.92	164.5

Source: IAP/DIBAP-ICMS Ecológico for Biodiversity.
Notes: Faxinais are traditional communities that exist in the center-south area of Paraná. APP, RL, SE and OFC, categories are only credited to the ICMS-Ecológico in buffer zones surrounding integrally protected areas, with the objective of connecting vegetation fragments.

Both in Paraná and Minas Gerais, however, a somewhat opportunistic effect associated with the creation of new municipal Environmental Protection Areas (APAs) was evident, which were not really effective in terms of biodiversity protection. Hence, for the assessment of conservation effectiveness of ecological fiscal transfers it is important to report which categories of protected areas increase, and that quality of these areas is assured. Although conservation factors for the calculation of ecological fiscal transfers provide an economic incentive for protected areas with higher conservation values and associated land-use restrictions, it may be easier for municipalities to increase municipal budgets by way of designating a large proportional area in Environmental Protections Areas with low land-use restrictions associated.

2.3 Cost-effectiveness of ecological fiscal transfers

Cost-effectiveness of conservation policies can be looked at from two different perspectives: 1) A conservation policy is more cost-effective than another if 1) its conservation outcome is higher for given total costs or 2) an equal conservation outcome is attained at lower total costs (Wätzold et al., 2010). Following Birner and Wittmer (2004) and Wätzold and Schwerdtner (2005), the total costs of conservation policies may be divided into production costs and transactions costs. Transaction costs may again be subdivided into implementation costs and decision-making costs.

Productions costs are defined as the costs of actual conservation measures. These are not relevant for ecological fiscal transfers, because this instrument usually does not aim to directly finance conservation measures. The primary aim of ecological fiscal transfers is to compensate the relevant jurisdictions for the land-use restrictions imposed by protected areas that in economic terms relate to the opportunity costs of these protected areas. Due to this compensation the provision of the related public good 'protected areas' may or may not increase (there is no earmarking or contingency, except in Paraná due to quality assessment).

Implementation costs include the costs of introducing and implementing the policy instrument itself, in our context the costs associated with introducing new ecological indicators into intergovernmental fiscal transfers and the associated monitoring and enforcement tasks. In the case of ecological fiscal transfers, the necessary institutions are already present. Therefore, introduction costs for ecological fiscal transfers are reasonably low. This holds especially true if easily available indicators are used such as the protected area coverage (Ring, 2008a). If a quality criterion is implemented, like in the case of Paraná, the quality of protected areas needs to be monitored at regular intervals by conservation authorities.

Decision-making costs relate to the costs of acquiring the information necessary for the successful design and implementation of conservation measures. This includes knowledge on the natural resources, information on preferences in the case of conflicting goals and information on production costs (Birner and Wittmer, 2004). They also include the costs of coordinating decision-making if different individuals or stakeholder groups are involved. They may include the resources spent on meetings and resolving conflicts, for example. Again, this category of transaction costs is of minor relevance for ecological fiscal transfers, especially it the easily available indicator of existing protected area within municipal territory is used as a basis for allocation of fiscal transfers.

To sum up, productions costs do not apply to ecological fiscal transfers, if the aim of the instrument is to compensate for opportunity costs only. Transaction costs of ecological fiscal transfers are comparatively low, because it does not require new institutions or a new bureaucracy (Ring, 2008a): by introducing an easily available ecological indicator into the existing fiscal transfer mechanism, such as protected area size, it builds on existing institutions and administrative procedures. This is not true however for the implementation of qualitative indicators, which requires a regular field validation of protected area management quality and relevance to local sustainable development. However, the effectiveness of the ICMS Ecológico is far greater with implementation of these parameters.

2.4 Social impacts

Intergovernmental fiscal transfers are core to fiscal and social policies and are usually used to equalise public revenues across rich and poor jurisdictions according to their fiscal capacities and needs. This opens an alley for fiscal transfers to combine conservation policies with poverty alleviation objectives (OECD, 2005), an important characteristic for designing policies in developing and transition countries. However, intergovernmental fiscal transfers are just a subsidiary instrument in intergovernmental fiscal relations. More important in terms of distributive effects is the general tax structure in a country and the primary sources of public revenues. Especially in Latin American countries, most public revenues still stem from value-added taxes that hit the poor harder than the rich. Therefore, distributive aspects primarily need to be dealt with as part of the general tax system of a country.

Having said this, one can principally investigate the social impacts associated with the introduction of ecological fiscal transfers. This analysis highly depends on the structure of the relevant fiscal transfer scheme and where exactly ecological fiscal transfers are introduced. Usually, with any change in indicators in fiscal transfer schemes, there are winning and losing municipalities. This is the reason why amendments to financial constitutions are such highly politicised processes and are often brought to constitutional courts.

At the outset, it is important to know whether new funds are available for the ecological component (e.g. protected areas) in the fiscal transfer scheme. If this is the case, local governments with protected areas increase their municipal revenues and belong to the winning municipalities of the new scheme, whereas others without or few protected areas more or less get the transfers they used to get before ecological indicators were introduced.

In the case of the ICMS Ecológico, a certain percentage of state ICMS revenues – the most important tax in terms of public revenues at state level – is reserved for distribution among local governments with conservation units. This clearly leads to winning and losing municipalities, because other indicators have been lowered with the introduction of the ICMS Ecológico. Nevertheless, the success of ecological fiscal transfers is linked to a strong tax, the revenues of which usually show an increasing trend over the years, especially in times of economic growth. Therefore, overall fiscal transfers as well as ecological fiscal transfers increase with rising state ICMS revenues. This may even counteract the effect of decreasing transfers per hectare protected area as newly created protected areas in a state draw from a fixed budget (in case the percentage growth of revenues matches the percentage growth in protected area).

In other countries, ecological indicators may also be introduced as part of the fiscal need determination of a jurisdiction which is then entered in a formula-based procedure and weighted against the fiscal capacity of the relevant jurisdiction. Although there are winning and losing municipalities with the introduction of a new ecological indicator, such a system may be considered socially equitable. Especially in transfer systems with a strong equalising component, ecological fiscal transfers are then also highly dependent on own-source public revenues. If, for example, own source revenues from land or business taxes are high (i.e. the richer a municipality), a municipality may in any case get little or even no transfers. In this case, protected area coverage must substantially increase the municipality's fiscal need in relation to its fiscal capacity to lead to increased transfers. Otherwise the jurisdiction is assumed to be capable to deal with the associated land-use restrictions (Ring, 2008b). However, if own source revenues are low and protected area coverage high, then the new ecological indicator increases fiscal need and thus the transfers received by the municipality. In this way, poorer municipalities in rural areas with low revenues (e.g. due to low population densities and/or little economic activities) and high protected area coverage benefit most.

Next to strengthening ecological indicators in fiscal transfer systems, a social or equity-related dimension may also be promoted. For example, in Minas Gerais, next to the ICMS Ecológico, social components were introduced into ICMS distribution from the state level to the local level of government. This is commonly referred to as the "Robin Hood Law" (Bernardes, 1999; Grieg-Gran, 2000), involving pro-poverty politics with low revenue municipalities receiving some of the richer municipality revenues based on indices related to relative poverty or GDP/capita. This latter aspect needs to be separated from the ICMS Ecológico aspect, which is focused primarily on the conservation share of the locality's area. Ecological indicators may reinforce the equity aspect of the allocation mechanism, if they adjust for quality aspects associated with bringing biodiversity benefits to disadvantaged groups. Nevertheless, there are limits to win-win strategies relating to ecological and social objectives. "Effective management" of parks and reserves that are meant for exclusion of humans may mean clamping down on social benefits derived from low impact uses of such lands by communities neighbouring on or residing within these areas. This is a major discussion and social conflict within the SNUC in Brazil and not just a conceptual issue. In Paraná, this has been partially offset by the ICMS-E payments for *faxinais* and some resources made available to agrarian reform beneficiaries who protect forest remnants.

2.5 Institutional context and legal requirements

Regarding formal institutions, ecological fiscal transfers are part of intergovernmental fiscal relations. Basic rules for intergovernmental fiscal relations are usually covered in a country's constitution that includes basic financial arrangements between levels of government. For federal systems it is common to have a constitution or relevant legislation at the national level that sets the frame for state level constitutions and laws. Therefore, to newly introduce ecological indicators into fiscal transfer systems at any level of government, some constitutional changes may be necessary.

For example in Brazil, Article 155 of the Federal Constitution specifies the ICMS as a state levy on the circulation of goods, services, energy and communication. The Federal Constitution also prescribes that 25 % of this largest source of state revenues in Brazil is to be allocated to municipalities. Of the latter share, each state can define the criteria to distribute 25 %, whereas 75 % are distributed based on an index of municipal economic output (May et al., 2002). At state level in Brazil, State Constitutions first need to be amended to enable the adoption of the ICMS Ecológico. The ICMS Ecológico is then introduced as a State Law, often followed by further implementing decrees. Some states such as Paraná move quickly in this direction with the State Constitution's change in 1989 and the ICMS-E Law in 1991, followed by the implementation of the scheme in 1992, when municipalities first received fiscal transfers for protected areas. Other states take substantial time for the various steps involved from constitutional change to implementing laws and regulations, as can be seen in the case of Acre, where constitutional change already took place in 2004, but implementing laws and regulations are still pending (cf. Table 1).

On the side of ecological indicators to be newly introduced into the fiscal transfer schemes, there are also some formal requirements. Usually, indicators in these schemes need to be sufficiently easy to grasp and monitor, and statistically available (Ring, 2008b). This is certainly a reason for existing ecological fiscal transfer schemes to introduce protected area coverage as a surrogate indicator for biodiversity conservation and nature protection. Many fiscal transfer schemes already use an area-related indicator for distributing transfers, thus it is only a small step to move towards a protected area-based indicator. These designated protected areas need to be officially registered or gazetted before municipalities are eligible to receive ecological fiscal transfers on their basis. In some cases, municipalities became aware of protected areas within their territory due to the introduction of the ICMS Ecológico (May et al., 2002).

For some municipalities with high protected area coverage, the fiscal compensations have been very significant. May et al. (2002) and Ring (2008a) indicate that fiscal transfers based on the ICMS Ecológico can amount to significant proportions of the overall municipal budget. The TNC (2010) webpage has published available data on municipal ICMS Ecológico revenues for the states of Ceará, Mato Grosso do Sul, Minas Gerais, Paraná, Rio de Janeiro and São Paulo. A primary issue not very well dealt within the literature is – what is the incremental money used for? May et al. (2002) empirically investigated this question for selected municipalities in Paraná and Minas Gerais. As the ICMS Ecológico is generally distributed as a lump-sum transfer, municipalities can use the money in any way they wish. Hence, the new revenues based on protected areas may even be used for harmful activities, and potentially destroy or degrade valuable habitats. With the quality clause in the state of Paraná, there is at least some basis for accountability, representing a conditionality of transfers on effectiveness.

Regarding actor constellations to introduce ecological fiscal transfers, constitutional changes require specific – usually larger – political majorities. Here, power relationships among the actors involved play a crucial role. Political parties, public actors, in the form of governments at various governmental levels, but also non-governmental organisations (e.g. conservation NGOs) and associations (of municipalities or districts) all play an important role in moving towards ecological fiscal transfers. These relationships may be based on competitive or cooperative behaviour and impede or facilitate new policy instruments. Box 2 summarises important lessons learned in terms of good governance and institutional context for ecological fiscal transfers, building on experience with ICMS Ecológico legislation in the Brazilian states.

Box 2. Good governance for ICMS Ecológico legislation

Wilson Loureiro, State Environmental Agency, Paraná (as interpreted by Peter May)

Considering the constitutional and legal possibilities, the author's accumulated experience justifies some observations in regards as to what can be considered a good quality statute on this theme, from its conception, to its being put into practice. Among these, the following stand out:

a) Establishment of space for dialogue with the various sectors within the State and with organised civil society;

b) It is no panacea: the ICMS Ecológico is an instrument that provides means but that does not in itself provide for its ends and that functions best when combined with other instruments;

c) The municipality is the linchpin in the national and state environmental system, and as such, it is reasonable to take advantage of these laws to nurture decentralisation of environmental management;

d) When environmental themes are defined (for qualitative appraisal), efforts should focus on resolving what is essential; it is not reasonable to imagine that the law can resolve everything in "one fell swoop", but rather to select a subset of criteria, that permits the ICMS Ecológico to be implemented;

e) Periods of amnesty and programmes of support and training of municipalities for adaptation, for the gradual and progressive implementation of the ICMS Ecológico, and flexibility in the models for improvement as time goes by are essential elements in its legal conception;

f) When discussing biodiversity conservation, it is recommended that besides conservation units, that the law contemplate indigenous peoples, quilombolas (lands occupied by descendents of slaves) and other traditional communities, and

g) stimulate opportunities for achieving connectivity between forest fragments, with the objective of creating biodiversity corridors.

h) Experience has shown that in the utilisation of quantitative indicators, made even more complex by the inclusion of qualitative criteria (such as the example of biodiversity conservation utilised in Paraná), that their measurement viability guide the selection of such indicators;

Finally, it is to be recommended that the Law be generic, that it attend to legal and constitutional precedent, that it provide statutory support to administrative procedures, but that above all that adjustment in the ICMS Ecológico law be enabled, permitting that solutions for priority environmental problems be achieved by associating this instrument with other mechanisms under development in each state. In other words, it is desirable to have a concise, tight enabling statute, but that it be flexible and adaptive.

Once, ecological fiscal transfers are introduced, they are automatically distributed. This may lead to the fact that municipality are not even aware of receiving transfers for conservation areas. May et al. (2002) have analysed responses on the ICMS Ecológico in Paraná and Minas Gerais. State conservation (Paraná) or forest agencies (Minas Gerais) developed information policies to let stakeholders know about the new instrument as a prerequisite to spark off any incentive effect towards more conservation. Some municipalities engaged in actively informing their inhabitants about the new revenues based on conservation areas and they report on the uses of the monies.

Last but not least it is important to state that ecological fiscal transfers are but one (economic) instrument in the mix of relevant policy instruments for biodiversity conservation. More about the role of ecological fiscal transfers in a policy mix follows in a later section.

3 The new Local Finances Law in Portugal

3.1 Promoting sustainable development and biodiversity conservation

Within the European Union, Portugal is a pioneer in introducing ecological fiscal transfers. Portugal is the first EU Member State to recognise protected areas as an indicator for the redistribution of public revenues through fiscal transfers from national to local governmental level (Santos et al., 2010). In January 2007 a new scheme of fiscal transfers for biodiversity conservation was introduced with the approval of a revised Portuguese Local Finances Law (LFL – Law 2/2007, 15th January). This law establishes the general principles and rules for the fiscal transfer from the national to the local governmental level in Portugal. The LFL of 2007 introduced a new Article 6, dedicated to the promotion of local sustainability. This article establishes that 'the financial regime of municipalities shall contribute to the promotion of economic development, environmental protection and social welfare'. This objective is supported by various mechanisms, including positive discrimination for those municipalities with land designated as Natura 2000 network or other national protected areas (Santos et al., 2010). For the first time, conservation areas affect the allocation of funds from the General Municipal Fund (FGM – Fundo Geral Municipal) and this mechanism effectively constitutes an ecological fiscal transfer.

The LFL specifies three different funds for the transfers from the national to the local level, the Financial Equilibrium Fund (FEF), the Municipal Social Fund (FSM) and a variable fraction corresponding up to 5 % of the IRS collected from individuals living in the municipality. The FGM is equal to 50 % of the FEF; the remaining 50 % of the FEF is allocated to the Municipal Cohesion Fund (FCM), whose aim is to balance out levels of development and opportunities among municipalities. FGM moneys are allocated to municipalities according to the following criteria: 5 % is distributed equally to all municipalities, whereas 65 % is allocated as a function of population and of the average number of stays in hotels and on campsites. The remaining 30 % is allocated on an area basis, including a differentiation between general area and conservation area. In municipalities with less than 70 % of their territory under designated conservation areas, 25 % is allocated in proportion to the area, weighted by elevation levels, and 5 % in proportion to land designated as Natura 2000 or other nationally protected areas. In municipalities with more than 70 % of their territory under conservation regimes, 20 % is allocated in proportion to the area, weighted by elevation levels, and 10 % in proportion to land designated as Natura 2000 or other protected areas. As a consequence, municipalities with higher land-use restrictions (more than 70 % of their territory designated as conservation areas) receive higher transfers per hectare conservation area than municipalities with less than 70 % of their territory under conservation regime (see Table 4).

3.2 Fiscal effects of protected areas on municipal budgets in Portugal

The step to consider protected areas in the new Portuguese LFL happened very recently. Therefore it is not yet possible to observe a change in protected area coverage, as a potential incentive effect of the new law in terms of environmental effectiveness. Santos et al. (2010) analyse the new Local Finances Law and compare it with its predecessor, highlighting changes in fiscal revenues for selected municipalities in the country in relation to their designated protected areas.

Any changes in the LFL allocation criteria are very important in terms of funding and, consequently, the development strategy of municipalities with a high dependency on fiscal transfers. In municipalities such as Barrancos, 97 % of total municipal revenues come from fiscal transfers. Amendments to the LFL undertaken in 2007 relate to various funds and allocation criteria. For this reason, there are several crossover effects that have significant implications for the final allocation of funding to each municipality. In addition to providing an analysis of the overall financial effects and the incentive impact of the new law, Santos et al. (2010) have analysed its effects for a selected sample of 26 municipalities in terms of its ecological component by considering two different simulations: 1) the new law as it stands compared with the estimated fiscal transfers for 2008 applying the old LFL criteria (considering two variants) and 2) the new law as it stands vis-à-vis the new law without the ecological component. These aspects were analysed on the basis of data from 2008.

1a) To assess the effects of the new LFL, the real values of the 2008 fiscal transfers were compared to the estimated fiscal transfers for the same year applying the old LFL criteria (including the criteria for calculating the total national transfer value for 2008) and the criteria for allocation among the municipalities. With this analysis it is possible to identify which municipalities win and which ones lose out as a result of the changes in the law. This analysis showed that total fiscal transfers would be 5.5 % higher if the old law was still applied. Thus, the changes introduced by the new LFL reduced the total amount transferred from the national level to municipalities. In fact, all the municipalities included in the sample lose out as a result of the changes in the law (Santos et al., 2010).

1b) In order to eliminate the effect associated with the differences in the amount of total fiscal transfers, Santos et al. (2010) developed an alternative scenario which assumed an equal total amount in the application of both laws (the real value for 2008 resulting from the new law). This approach allowed isolating the effects resulting from differences in the allocation criteria for the funds. A comparison of this new scenario for the old law with the new LFL showed that only 62 % of municipalities lose out under the new law (16 out of 26), whereas in the previous case 1a) they were all losers. The losses are also less significant (5.9 % at most). Ten municipalities win with the new LFL funds allocation criteria, eight of them belonging to the group with less than 70 % conservation status area. This result indicates that the introduction of the ecological component was not sufficient to counterbalance other effects and provide a greater incentive to those municipalities with a larger proportion of protected areas.

2) Santos et al. (2010) developed a second scenario to illustrate the situation that arises when the criterion “area” does not include the proportion of the municipality’s land that has conservation status. It is assumed that 30 % of the FGM is assigned to each municipality according to area (weighted by elevation levels) with the remaining 70 % of the FGM to be allocated without any changes. This simulation isolates the effect achieved by introducing the ecological criterion in Portuguese fiscal transfers. The results “show that all municipalities with more than 70 % of their territory under Natura 2000 or protected areas regimes would lose out if the new LFL was applied without the ecological criterion. In the group of municipalities with less than 70 % of their territory under Natura 2000 or other protected areas regimes, there are 9 winners and 6 losers, but those that lose out have a relative loss lower than that for the other group of municipalities. In the first group the average loss is -15.2 %, while in the second group the average loss is only -1.4 %.” (Santos et al., 2010).

Finally, Santos et al. (2010) present the percentage share of the ecological component as a proportion of total municipal revenues and fiscal transfers (Table 4). This share is significant for the municipalities in the group with more than 70 % of designated area (between 4 % and 38 %) and much higher on average than in the group with less than 70 % designated conservation area (less than 8 %). At either extreme along this scale are Castro Verde, where the ecological component is 38 % of total fiscal transfers and 34 % of total municipal revenue, and Lisbon, Almeirim and Aguiar da Beira, where the ecological component is zero due to completely missing conservation areas of the latter.

Table 4. The Portuguese new Local Finances Law: Share of the ecological component in municipal revenues

		Share of the ecological component						Ecological component per unit		
	Municipalities	Total municipal revenues	Fiscal transfers	Estimated ecological component (euros)	Population (inhabitants)	Total municipal area (ha)	Designated conserva-tion area (ha)	€/in-habitant	€/ha of total municipal area	€/ha of CA
Municipa-lities with *more* than 70% CA	VILA DO BISPO	13%	22%	873,332	5,423	17,900	17,423	161	49	50
	PORTO MONIZ	9%	9%	353,343	2,706	8,300	7,049	131	43	50
	MURTOSA	6%	8%	294,729	9,804	7,300	5,880	30	40	50
	PORTO DE MÓS	11%	15%	1,002,546	25,022	26,200	20,000	40	38	50
	RIBEIRA BRAVA	4%	5%	250,733	5,349	6,500	5,002	47	39	50
	ALJEZUR	16%	22%	1,191,281	12,565	32,400	23,765	95	37	50
	BARRANCOS	26%	26%	843,298	1,767	16,825	16,823	477	50	50
	CAMPO MAIOR	25%	28%	1,238,105	8,342	24,700	24,700	148	50	50
	TERRAS BOURO	22%	23%	1,318,523	7,765	27,700	26,304	170	48	50
	FREIXO E. CINTA	21%	23%	1,110,681	3,931	24,400	22,157	283	46	50
	CASTRO VERDE	34%	38%	2,167,498	7,772	56,900	43,240	279	38	50
Municipa-lities with *less* than 70% CA	LISBOA	0%	0%	0	509,751	8,500	0	0	0	0
	LAGOA	0.1%	0.1%	2,698	15,139	4,600	108	0.2	1	25
	GRÂNDOLA	2%	3%	173,582	14,214	80,800	6,926	12	2	25
	SINTRA	0.3%	1%	286,077	428,470	31,900	11,414	1	9	25
	VIANA CASTELO	0.5%	1%	120,256	73,559	19,700	4,798	2	6	25
	AMARANTE	1%	1%	205,889	91,238	31,900	8,215	2	6	25
	SESIMBRA	2%	5%	259,978	61,471	30,100	10,373	4	9	25
	AVEIRO	1%	3%	240,676	48,110	19,500	9,603	5	12	25
	ALMEIRIM	0%	0%	0	22,766	22,200	0	0	0	0
	AGUIAR BEIRA	0%	0%	0	6,262	20,700	0			
	PESO DA RÉGUA	0.4%	0.5%	28,369	17,492	9,500	1,132	2	3	25
	LAMEGO	1%	2%	136,491	26,484	16,500	5,446	5	8	25
	ÉVORA	1%	1%	192,472	55,420	48,200	7,679	3	4	25
	COVILHÃ	2%	3%	389,338	52,946	130,704	15,534	7	3	25
	VIMIOSO	8%	8%	522,381	4,975	55,600	20,842	105	9	25

Legend: CA: Conservation area (Designated Natura 2000 areas and nationally protected areas)
Source: Santos et al. (2010).

The analysis presented by Santos et al. (2010) shows that ecological fiscal transfers can be significant for some municipalities in which the amount of land granted conservation status constitutes a large part of their overall territory. The ecological criterion works as an incentive for municipalities to maintain or increase their protected areas, in terms of their quantity, however, this must be complemented with quality criteria to also incentive the management of those areas. The overall reduction in the global value of fiscal transfers (when compared with the amounts that would have been transferred if the law had not been changed), combined with the crossover effects of changing various indicators for distribution, however, has contributed to lessening the financial incentive offered to municipalities through ecological fiscal transfers.

4 Suggested ecological fiscal transfers schemes

In other countries, conservation-based indicators have been recommended by environmental expert commissions (e.g. in Germany: SRU, 1996) or proposed as an option in the scholarly literature, in some cases accompanied by spatially explicit modelling of the potential fiscal consequences for local governments. Spatially explicit modelling and GIS tools can help to illustrate the consequences of ecological fiscal transfers where they have not yet been introduced. Fiscal transfer schemes are country-specific and highly politicised due to the substantial financial flows involved. Building on existing fiscal transfer schemes and integrating suitable ecological indicators can help decision makers in promoting innovative and sustainable solutions.

For Switzerland, Köllner et al. (2002) developed a model based on biodiversity indicators and cantonal benchmarking for intergovernmental fiscal transfers. Building on empirical reviews of already existing ecological public functions in German state fiscal equalisation laws (Ring, 2002), Ring (2008b) developed and applied a protected area-based indicator (so-called conservation units) for integrating nature conservation into intergovernmental fiscal transfers (cf. Box 3). Taking as an example the fiscal transfer system at the local level in Saxony, a federal state (Land) in Germany, the fiscal impacts for Saxon municipalities of using this indicator have been modelled in a spatially explicit way for the fiscal year 2002, presenting two different options for using this indicator. The first option integrated conservation units into the calculation of general lump-sum transfers, as an additional indicator next to inhabitants and schoolchildren to determine the fiscal need of the relevant municipality in Saxony (cf. Box 3). As part of the second option (not depicted in Box 3), a certain amount of money was set aside and distributed to municipalities based on conservation units in relation to the total municipal area. It was assumed that no additional monies were available; therefore the money set aside reduced the amount available for general lump-sum transfers. In principle, both options are suitable for including protected areas in intergovernmental fiscal relations in the German state of Saxony. As a result, both options lead to higher transfers to rural and remote areas with high conservation value and less fiscal transfers to urbanised regions with little conservation areas. Whereas the second option leads to ecological fiscal transfers to all municipalities with protected areas, the first option compares increased fiscal need due to protected areas with the fiscal capacity of each municipality. If fiscal capacity exceeds fiscal need, the municipality receives no (ecological) fiscal transfers, assuming that the municipality copes well with the land-use restrictions imposed by protected areas.

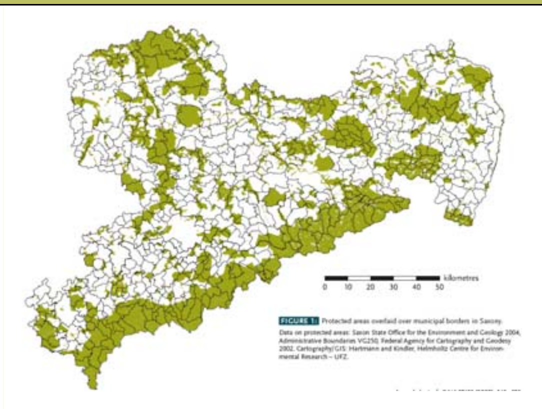
In contrast to the easily available indicator suggested by Ring, Perner and Thöne (2005) suggest a so-called “eco-points” approach for Germany, based on landscape planning procedures and relevant indicators. However, landscape plans – an additional feature in German local and regional land-use planning addressing nature conservation and landscape protection – are not even available for all municipalities in Germany.

For Indonesia, the extension of the formula to calculate the fiscal need of a jurisdiction has been suggested to account for conservation. Thus far, population, regional economic growth, a Human Development Index and a general area indicator are used for calculating the fiscal need of provinces or municipalities. Mumbunan (2011) and Mumbunan et al. (2010) suggest a protected area-based indicator for inclusion into the intergovernmental fiscal transfer scheme and model fiscal consequences for equalising transfers from the national to the provincial level. Kumar and Managi (2009) suggest the integration of forest cover into the countries intergovernmental fiscal transfer scheme. In Norway, the new Nature Diversity Law of 2009 mandates compensation for costs of establishing land-use regulations for biodiversity conservation. White papers evaluating the implications of the law discuss the possibility of using fiscal instruments (Norwegian Department of Finance, 2009).

In the near future, ecological fiscal transfers may also play a role in the implementation of international programmes on a nationwide scale, linking climate mitigation with biodiversity conservation policies (Ring et al., 2010). In Indonesia, for example, many local governments perceive forest exploitation and land-use change to be among the easiest ways to generate local public revenues (Barr et al., 2006), because local budgets benefit from logging activities within municipal boundaries. Forest conservation, by contrast, does not add to the budget. Therefore, REDD and REDD+ initiatives (Reducing Emissions from Deforestation and forest Degradation) will need to take into account fiscal transfer schemes to the local level as one important means of channelling international payments for biodiversity conservation and climate mitigation from the national down to lower levels of government, thereby contributing to the successful national implementation of international REDD and REDD+ schemes (Irawan and Tacconi, 2009; Ring et al., 2010).

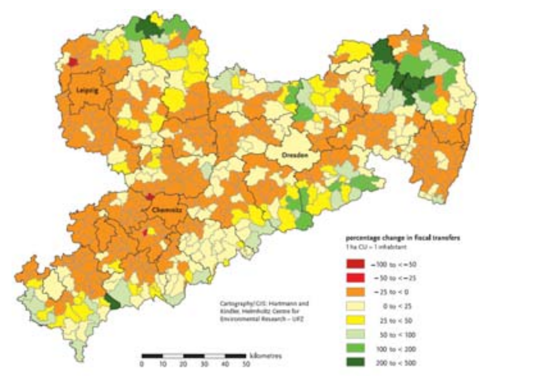
Box 3. Modelling intergovernmental fiscal transfers for biodiversity conservation in Germany

This model of Saxony's fiscal transfer system from state to local level is based on administrative, social and economic data from 2002. It has been enlarged by the conservation units (CU) indicator to take account of local ecological services whose benefits cross municipal borders. CUs are standardised areas within the borders of a municipality that belong to existing categories of Protected Areas defined by Saxony's Nature Conservation Law (a). The map in (b) shows relative changes in general lump sum transfers if CUs are used in addition to existing indicators (inhabitants and schoolchildren) to calculate the fiscal need of a Saxon municipality (Ring, 2008b).



(a) Protected areas overlaid over municipal borders in Saxony, Germany.

Source: Ring, 2008b.



(b) Percentage change in general lump-sum transfers when the Saxon fiscal transfer system 2002 was expanded to include designated protected areas.

5 The role of ecological fiscal transfers in a policy mix

Ecological fiscal transfers for biodiversity conservation build on existing protected area regulation in that they use officially designated protected area as an indicator to allocate fiscal transfers. In this way, they complement conservation law with an economic instrument that accounts for the local conservation costs and spillover benefits related to these protected areas. Ecological fiscal transfers explicitly address public actors, i.e. governments at different governmental levels and related public authorities. In this way, they also complement programmes and schemes that primarily address land users and thus, private actors in their conservation costs.

However, in practice we find ecological fiscal transfers at the local level in countries where traditionally, comparatively less programmes exist or used to exist directly addressing land users (Ring, 2008a). This is at least the case for Brazil, where fiscal transfers for biodiversity conservation have existed for some time. To the contrary, in many European member states, but also Latin American countries, a number of payments for environmental service schemes (e.g., Costa Rica's PES scheme) or agri-environmental or conservation support programmes as in the European Union member states exist, compensating land users for their costs of providing conservation-related services to society. With the recent exception of Portugal, these latter countries have not yet integrated nature and biodiversity conservation into intergovernmental fiscal transfers, although discussion to do so is going on for quite a while in some of these countries (e.g. for Germany, SRU, 1996; Ring, 2002; Perner and Thöne, 2005). Ideally ecological fiscal transfers addressing public actors and PES programmes addressing private actors should complement each other. Each of these actors encounters specific costs in providing conservation-related services. So there is a space yet to be determined where both these types of economic instruments make sense and complement each other.

Overlap of instruments may occur if there are conservation support programmes where local governments are also eligible to apply. These programmes then correspond to specific purpose grants. However, if in this specific case ecological fiscal transfers are mainly used to account for opportunity costs of biodiversity conservation in terms of lost tax revenues for local governments and other programmes to provide the management costs for conservation measures, then overlap can be avoided.

Ecological fiscal transfers have now existed for almost twenty years in Brazil. Although the instrument was designed to compensate municipalities for opportunity cost encountered with existing protected areas, it developed into an incentive to engage in the management of existing protected areas and to designate or support new ones (May et al., 2002; Loureiro, 2008; Ring, 2008a). This exemplifies the fact that protected area regulations as typical command-and-control instruments on their own are not enough. Instead, a combination of regulation and economic instruments capable of offsetting the costs associated with protected areas is required; such a linkage creates synergies and enables the spillover benefits generated to be internalised, at least to some extent. The increased supply of biodiversity conservation in the relevant Brazilian states through more and better managed protected areas could only feasibly be achieved at higher social cost by means of protected area regulations alone (Ring et al., 2010), even though the situation significantly differs between the relevant states. The remaining biodiversity in e.g., the Atlantic Forest lies outside of protected areas, and there is little prospect that such lands would become protected areas without expropriation for the public good. This is not the same in the Amazon, where most of the remaining forest is in poorly protected public land. In this situation, it is of interest to exploit the possibilities of APAs and RPPNs as a complement to non-use protected areas, even if the former has been abused to some extent where the quality criteria are not yet implemented.

All in all, ecological fiscal transfers have the potential to turn the oft-encountered local opposition towards protected areas into active support, but for this to occur requires that municipalities and/or state governments inform the citizenry and local officials of the relation between protected areas and the additional revenues, and make an effort to reward them for enhancement in biodiversity protection in an adaptive governance strategy.

6 Conclusion

Intergovernmental fiscal transfers have long been used to support local governments in providing public goods and services. To date, only Brazil and more recently Portugal, have implemented fiscal transfers for biodiversity conservation, acknowledging protected areas as an indicator to allocate fiscal transfers from higher governmental levels to municipalities. Although originally introduced as a compensation for local opportunity costs associated with the land-use restrictions imposed by protected areas, in Brazil, ecological fiscal transfers have developed as an incentive to designate more protected areas.

With regard to effectiveness for biodiversity conservation, the design of the conservation-related indicator is important. In most cases, only the quantity of protected areas is used for determining ecological fiscal transfers. Brazilian states also introduced a conservation factor that weighs the protected areas according to management categories. The higher the land-use restrictions associated with the protected area category, the higher the conservation factor and thus the fiscal transfers received. Despite such a weighing factor to favour protected areas with high conservation value, it may still be easier for governments to increase municipal budgets by way of designating large amounts of protected areas with low land-use restrictions which are not really effective in terms of biodiversity protection. This should be considered in the design of ecological fiscal transfers, e.g., through appropriate weighing of the relevant management categories or exclusion of areas with very low conservation value. Additionally, not just the quantity, but also the quality of protected areas should influence the amount of ecological fiscal transfers received. However, this requires a regular and reliable monitoring programme and leads to increased transaction costs, if monitoring of protected areas still has to be set up.

Concerning the role of ecological fiscal transfers in a policy mix, they are clearly addressing public actors at various governmental levels. Fiscal transfers for biodiversity conservation build on protected area regulation, compensating for the land-use restrictions and opportunity costs of local governments associated with these areas.

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PES and other Economic Beasts:
Assessing PES within a Policy Mix in Conservation

Ina Porras, Adriana Chacón-Cascante, Juan Robalino and Frans Oosterhuis

Summary

Payments for Environmental Services (PES) are transfers made directly to landholders to compensate them for good land management, including conservation activities. This mechanism has become one of the most popular market-based instruments used around the world. There are examples of important PES schemes in Asia, Europe, Latin America and Africa and as such, the instrument has been widely studied and analysed. Some of these assessments and the lessons learned are reported in this chapter concentrating on the role of PES within a policy mix framework as the instrument is seldom applied in isolation but rather as a component of complex policy arrangement combining different policy instruments.

1 What are payments for environmental services?

1.1 Definition and key features

Economic instruments play an important role in protecting biodiversity and other ecosystem services. In theory, they provide a more efficient and cost-effective option to traditional command and control tools like prohibitions and creation of protected areas. This chapter discusses the merits of one specific policy instrument, Payments for Environmental Services (PES), from theory to how it fares in the practice, as a stand-alone instrument and as part of a policy mix.

At its basic form, Payments for Environmental Services (PES) are transactions that reward those who look after ecosystems which benefit others (Bond and Mayers, 2010). The core of mainstream conceptual basis for PES is Coasean economics, where markets, or market-based forms, are used to promote bargaining initial allocation of property rights in order to achieve socially optimal levels of environmental externalities. The main goal of the instrument is to make conservation more attractive to landholders (or service provides) from an economic perspective by internalising the positive externalities attached to conserved resources such as water provision, carbon sequestration, and biodiversity conservation among others (Engel et al., 2008).

A more or less agreed definition of PES is based on five main features (Wunder, 2005). It must be a *voluntary transaction*, where a *well-defined environmental service* (or a land use likely to secure it) is bought by a minimum of one service *buyer*, and sold by a minimum of one service *provider,* who has legal or de facto control over the habitat or land for the duration of the contract; if and only if the *provider secures service provision.*

In practice few PES schemes incorporate the five features, and more relaxed definitions have been adopted. For example, Muradian et al. (2010) regard PES as “a transfer of resources between social actors, which aims to create incentives to align individual and/or collective land-use decisions with the social interest in the management of natural resources”. Although useful, this definition leaves aside what is known as the conditionality condition, which separates PES from other conservation-promoting instruments.

For the purpose of this chapter, we define PES as the instrument that addresses environmental externalities through a variable payment. It involves the interaction of at least two agents: the **provider or seller** of environmental services, who responds to the offer of a payment (from a private company, NGO, local or central government agency); and, the **users** of the provided services, who can be distinguished from the seller and are the one making payments that ensures the environmental services are to be delivered. Participation in the PES scheme must be voluntary on the supply side and the payment must be conditional on previously agreed land use that is expected to provide an environmental service.

1.2 Ecosystem services targeted

Most PES schemes tend to concentrate around the ecosystem services provided by **biodiversity** (including several biodiversity deals, the US conservation and wetland banking and Australian BioBanking), **water quality and quantity available (**including regulatory-formed markets such as the US EPA Water Quality Trading, the US State Nutrient Trading Programs, the New South Wales salinity credit markets in Australia and voluntary self-organised cases such as Perrier Vittel, now Nestle Waters, in France); **and air quality** (markets related to SO_2, NO_X, carbon/greenhouse gases, including EU ETS and REDD) (Waage and Steward, 2007).

In developing countries, PES programmes have not focused on the ecosystem service per se, but on different types of land practices (some examples provided in Table 1) that are expected to provide a particular environmental service.

Sometimes, activities supporting the provision of bundled environmental services can be complementary. For example, conservation of forests helps preserve existing water quality and quantity, protects biodiversity, and increases landscape beauty. On the contrary, other projects aiming at incentivising provision of a given ecosystem service (ES) harm other important ES. For example, planted forest (representing 7% of all forest worldwide (FAO, 2010), aiming at selling carbon credits can result in reduced water flows and reduce biodiversity, especially in water-strained environments, and when using fast-growing, exotic species.

Table 1. Examples of land use and ecosystem services targeted

Land use promoted	Ecosystem Service	Example
Conservation and protection of existing ecosystems	Bundled	• Costa Rica: payments made per hectare per year for conservation, and represent most of the demand for PES in the country (Porras, 2010; Sánchez-Azofeifa et al., 2007) • A compensation programme in Finland provides incentives for the creation of new nature reserves providing habitats of threatened species or of great natural beauty (Tikka, 2003) • Voluntary forest conservation contracts in Norway (Barton, 2010; Skjeggedal et al., 2010) • Sweedish Nature conservation agreements (Naturvårdsavtal), normally signed for 50 years (EEA, 2010; Mayer and Tikka, 2006) • Austrian Natural Forest Reserve Programme (launched in 1995) which compensates for not harvesting for a period of 20 years
Agricultural practices: aim at providing environmental services and on-site economic returns to farmer.	Usually biodiversity and water	• Silvopastorial projects in Colombia, Nicaragua, and Costa Rica (Casasola et al., 2009; Ibrahim et al., 2010; Montagnini, 2009) • Organic agricultura in Costa Rica: National Electricity Institute (ICE) Project in La Angostura Dam • Agroforestry contracts in the PSA Programme in Costa Rica, Sumberjaya in Indonesia, and Jesus de Otoro in Honduras • Best management contracts in the Catskill-Delaware Watershed in New York • Most European countries use subsidies for agricultural ecosystems conservation funding, co-financed by the EU'S Common Agricultural Policy (CAP), for example the High Nature Value areas promoted by the in Europe1 (EEA, 2010)
Reforestation for commercial purpose (medium to long-term schemes with timber as main objective)	Usually carbon but also for watershed protection	• Six national PES schemes and approximately 11 small local watershed schemes promote reforestation (Porras et al., 2008) • Community reforestation contracts through Plan Vivo in Mexico, Uganda, Mozambique and other countries (www.planvivo.org) • REDD projects (Bond et al., 2009)
Rehabilitation of degraded ecosystems for protection	Biodiversity and water	• Removal of alien tree species in the Working for Water in South Africa • PCJ in Brazil to restore riparian forests (Porras et al., 2008)

For many other examples see: Baylis et al. (2008); Ferraro (2009); Landell-Mills and Porras (2002); Porras et al. (2008); ten Brink et al. (2011); Waage and Steward (2007); Wätzold et al. (2010)

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

PES have been used at different governance levels, depending on the nature of the ecosystem service and the reach of the externality (see Table 2 for examples). These include local, regional (or provincial), national, and international schemes.

Table 2. Examples of ecosystem services at various governance levels

Level	Mechanism	Environmental service	Example
International	REDD, CDM	Carbon sequestration	Fondo Bioclimático, Chiapas, Mexico (Brown et al., 2004)
	International donor transfers	Bird habitat protection	Los Negros, Bolivia (Asquith et al., 2008)
Regional	Trust Fund	Protection of water quality for human use	Fondo ProCuenca, Quito Ecuador
National	Tenders/Auctions	Biodiversity, cultural services	Northeim, Germany (Bertke and Marggraf, 2005)
Local	Conservation of existing forest and riparian habitat restoration	Protection of water quality through reduced sedimentation	Platanar Hydroelectric, Costa Rica (Porras et al., 2008)
	Tenders/Auctions	Salinity control	Wimmera Auction for Salinity Outcomes (Whitten and Shelton, 2005)
Mixed levels: National, Regional, International	REDD payments to Amazon Fund conditional on national effectiveness assessment; local project support distributed by Amazon Fund manager, the Brazilian Development Bank (BNDES)	Principal carbon storage; Conservation of biodiversity	The Government of Norway's International Climate and Forest Initiative in Brazil (Miljøverndepartementet, 2010)

International agreements are those contracted by individuals, or institutions, from at least two countries, where one of them agrees to protect a specific ecosystem in exchange of compensation, generally in cash. These agreements can be one-off voluntary donations or special loans, business transactions, or are the result of intensive negotiations. There are several examples of international biodiversity conservation agreements, mostly in the form of direct donations for conservation in reserves, biological corridors, or to support the creation of national-level programmes. Examples include KFW in Costa Rica, conservation concessions promoted by Conservation International, and the World Bank's Silvopastoral project in Colombia, Costa Rica and Nicaragua. So far, there are no water related trans-boundary PES schemes actually in operation, as the nature of water governance is very difficult to address. More recently, the most visible example of international multi-level PES schemes is the Reducing Emissions from Deforestation and forest Degradation (REDD) scheme (Bond et al., 2009), which is discussed as a separate chapter of instrument review made by the POLICYMIX project (see Chacón-Cascante et al., this report).

National PES schemes redistribute national wealth by making direct payments or compensations to landowners. Although the line is blurred, the main difference with traditional subsidies is that, at least in theory, payments are conditional on performance and can be suspended if the landowner defaults. Also, instead of using prohibitions and fines to discourage bad land management, PES makes payments to individuals, either private or indigenous groups, for good behaviour. Funding for government-led PES comes most often from general budgets, but also from target fees and taxes (like the fuel tax in Costa Rica, or water taxes in Mexico). Examples of national schemes include Costa Rica and Mexico, Probosque programme in Ecuador, subsidies for forest conservation in Colombia, the Environmental Quality

Incentives Program (EQIP)[2] in the USA, and the State designed voluntary conservation programmes in Finland (Tikka, 2003), Norway (Barton, 2010) and the USA. To some extent, the 'agri-environmental' schemes under the EU's Common Agricultural Policy, and the conservation programmes under the 2002 Farm Bill in the USA can also be seen as PES schemes, although they do not always meet all features of PES mentioned above (e.g. a 'well defined' environmental service and the conditionality requirement) (Baylis et al., 2008).

Local level schemes are used especially for watershed services such as water quality, where the links between land use and environmental service tend to be more clearly perceived by stakeholders. Some of these local schemes are linked to national or international initiatives, for example, Cuencas Andinas in South America, Silvopastoral schemes in Central America, and RUPES in South East Asia (Porras et al., 2008), but many evolve on their own, and tackle a specific situation (for example, the Vittel watershed protection programme in Eastern France (Perrot-Maitre, 2006).

1.3 Stakeholders involved

In general terms, participants can be grouped into three broad categories: buyers (and non-paying beneficiaries), providers, and facilitators. Buyers can either be actual users of the environmental service (user-financed programmes), or groups acting in behalf of actual users and making the decisions regarding the programme. Sellers are those agents who are in a position of safeguarding the provision of ES, and intermediaries and facilitators are those who help bridge the gap (Engel et al., 2008; Porras et al., 2008; Smith et al., 2006):

- **Buyers of environmental services** can be direct users (in direct contracts), or grouped under a third party, usually the government or some form of other group like an NGO:
  - *User-financed schemes* include private users, public (e.g. local municipalities), quasi-public groups (i.e. hydroelectric projects), or NGOs (e.g. conservation groups, and it represents the ideal "Coasian PES'. In theory at least it is more efficient as buyers are more involved, evidence of impact, and feedback channels more likely, and contracts can be relatively easily re-negotiated (Pagiola and Platais, 2007).
  - In *Third-party schemes* acting on behalf of others efficiency may be compromised because buyers are less likely to be in direct contact with service providers, and more prone to political pressures and expectations. Effectiveness can be higher because of the potential of economies of scale. This third party can be the government, either through compulsory fees, or redistribution of existing funds generally imposed (e.g. on final water users) or through token consultation. Governments can also act as buyers of ES, e.g. buying carbon credits through over the counter (OTC) transactions, A third party can also be an international agency (private, public, NGOs, development agencies), either through donations, loans, voluntary contributions, or global markets generated through REDD payments. Can be as efficient as direct user-pay if conditionality is applied strongly.
- **Sellers of environmental services** are those agents who are in a position of safeguarding the provision of environmental services during the contract (or specifically the land-based activities expected to provide these services). They can be private landholders, informal occupiers of public lands, communal landowners, and in some cases government or NGOs managing protected areas:
  - *Private landowners/land users (i.e. agriculture on leased land)* are expected to have clear ownership of their land, with either land titles or undisputed informal possession rights. Can be

[2] *http://www.nrcs.usda.gov/programs/eqip/*

individuals or private conservation groups (reserves). This is more the case of schemes in developed countries.

- *Communal landowners* are farmers living in communal lands, for example indigenous reserves, and ejidos in Mexico. Similar to the case of private landowners, land ownership by the group has to be legally recognised.
- *Informal occupiers* of public lands are farmers living in public lands (often designated as national park but poorly enforced). In some cases farmers can have long-standing rights over land but are not compensated when reserves are formed (i.e. Cayambe Coca and Antisana Reserves in FONAG, Quito), or they live in these parks (i.e. the Maasin Watershed Rehabilitation Project in the Philippines).
- Some schemes work with government or NGO managing protected areas, usually along buffer areas of national parks. Although schemes where the government is both buyer and seller does not count as PES, but as an internal transfer.

- Intermediaries, or **facilitators** are those groups or institutions that help bridge the gap between suppliers and users of ES. They can either manage the schemes or provide ancillary services. A key to equitable PES is the ability to find an intermediary to group small providers, often dispersed, and keep transaction costs low. They include NGOs (international, national and local), donors, government groups, the academic sector, trusts and user associations.

1.4 Baseline and monitoring

A baseline is needed to understand what would have happened in the absence of the programme. Definition of the baseline has an enormous impact on the evaluation of any PES project. Three main options are often discussed: static, declining, and improving (Wunder, 2007):

- A static baseline assumes that the provision of the environmental service will remain constant in time without payment (Figure 1-a);
- A declining baseline is used to explain resource degradation assuming that it is part of a process (e.g. deforestation), and any slow-down or cease would qualify for additionality (Figure 1-b);
- An improving baseline is chosen when degradation of a given resource or environmental service is perceived as already reverting even before the introduction of PES (Figure 1-c).

From figure 1, it is clear that the selection of the type of baseline determines how additionality is measured. As it will be discussed in the following section, effectiveness and efficiency of PES programmes depends on how successful they are to avoid leakage, ensure additionality and minimise waste of resources. Both requirements depend on a proper definition of the baseline (Engel et al., 2008). PES programmes that focus on additionality are at risk of creating perverse incentives, as landowners could be tempted to increase deforestation before signing the contract to change the baseline to be used for compensation estimation. If this is not properly controlled, PES effectiveness can be overestimated (Alpizar et al., 2007; Casasola et al., 2009; Chomitz et al., 1999; Ibrahim et al., 2010; Montagnini, 2009; Wunder, 2007).

Monitoring and evaluation are activities needed to ensure conditionality. It helps to guarantee those selling that payments are made based on the actual provision, and to guarantee those buying that the service is delivered (Engel et al., 2008; Meijerink, 2007; Montagnini, 2009; Rojas and Aylward, 2003). Monitoring systems will depend on whether the PES programme is input-based or output-based.

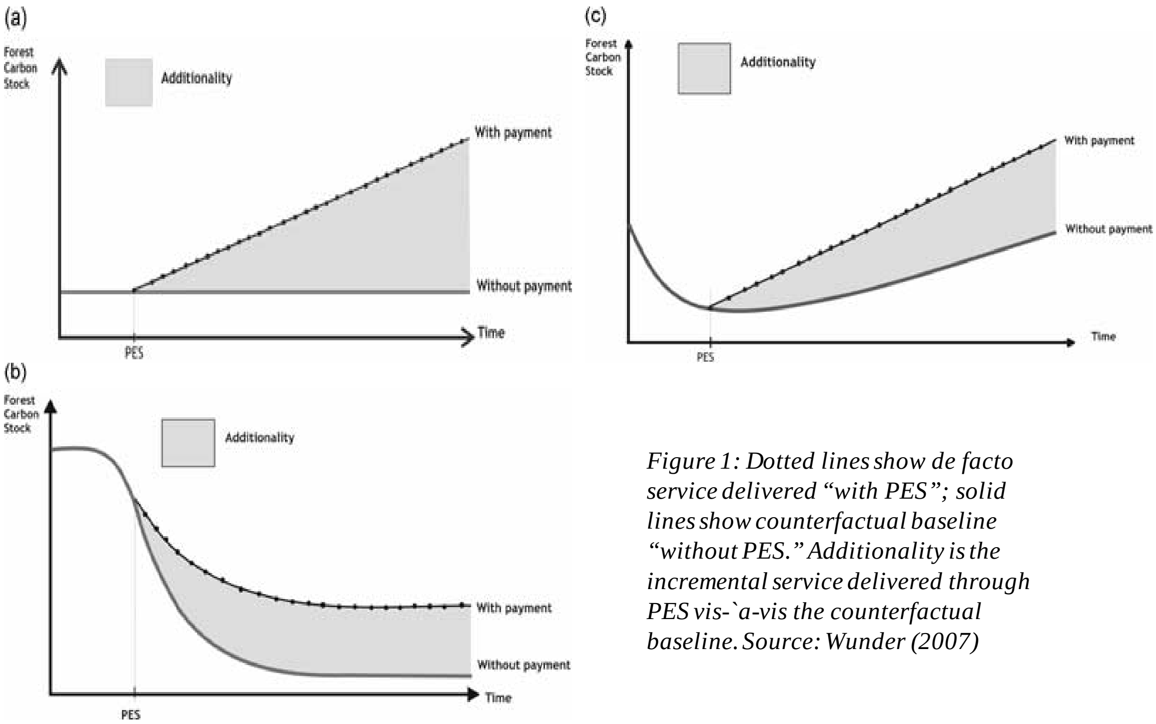

Figure 1: Dotted lines show de facto service delivered "with PES"; solid lines show counterfactual baseline "without PES." Additionality is the incremental service delivered through PES vis-`a-vis the counterfactual baseline. Source: Wunder (2007)

Figure 1. Different definitions of baselines for PES

Output-based programmes monitor the delivery of the environmental service through indicators (for example, tons of carbon sequestered or number of bird species protected). Input-based systems make assumptions on the impacts of predetermined land-use activities, and monitor these changes accordingly. Most ongoing schemes focus on compliance of land uses (Engel et al., 2008). Generally, monitoring costs are lower if only land-use compliance is checked. For this reason, for instance the Netherlands has recently removed the output-based elements (such as numbers of meadow birds) from its agri-environmental subsidy scheme.

2 Literature review of instrument performance

The underlying assumption of Coasean economics is the theoretical achievement of win-win situations. In the practice this is difficult (Barrett et al., 2005; Corbera et al., 2007a; Grieg-Gran et al., 2005), and efficiency needs to be addressed together with institutional and equity considerations, generally involving trade-offs, although the extent of these is still largely unknown because of lack of adequate information on impact evaluation. Although relatively new in the practice, PES has been subject to many critics (Agarwal et al., 2007; Alpizar et al., 2007; Asquith and Vargas, 2007; Blackman and Woodward, 2010; Börner et al., 2010; Casasola et al., 2009; Corbera et al., 2007b; Porras et al., 2008):

I. Lack of clearness in the relationship between some land-use practices and its impact on the environmental services they provide;

II. Difficulty in properly measuring the ecosystem service;

III. Unclear use of a market approach as opposed to subsidies;

IV. Carelessness in the approach of equity and poverty issues;

V. Lack of attention to institutional preconditions, such as land grabbing, insecure tenure, overlapping claims, and lacking information on private tenure;

This section presents existing literature discussing some of these issues, focusing as well on effectiveness, cost-effectiveness, efficiency, reported impacts from ongoing schemes, and the institutional context in which PES emerges and develops.

2.1 Measuring and valuing

A recurrent critic to PES focuses on the lack of clearness in the relationship between some land-use practices and their impact on the environmental services, and the difficulty of measuring this relationship. Although the lack of clarity between land use and conservation applies to any economic instrument used, it is particularly important for PES because of the way it affects the definition of the baseline and what should be considered additional (Agarwal et al., 2007; Alpizar et al., 2007; Asquith and Vargas, 2007; Casasola et al., 2009; Chomitz et al., 1999; Corbera et al., 2009; Echavarria et al., 2003; Engel et al., 2008).

The science underpinning carbon sequestration from forests has received major attention, fuelled by the magnitude of the numbers it invokes. Forest loss, primarily tropical deforestation and forest degradation, accounts for approximately 17 per cent of global greenhouse gas emissions (Rogner et al., 2007). Carbon contracts are either linked to carbon sequestration deriving from the net absorption of carbon dioxide in planted trees, or by protecting carbon stocks (which would otherwise be emitted if deforested) in natural forests (Bond et al., 2009). The sequestration value of a given forests or land plot is conditional on tree density and variety. As in the case of biodiversity conservation, other practices than reforestation and forest conservation can potentially enhance (or reduce) carbon sequestration. For example the GEP/silvopastoral project found that in Esparza, Costa Rica degraded pastures carbon banks were of just 26.4 tC per hectare compared to a range of between 114.4 and 143.0 tC per hectare found in improved and natural pastures with trees per hectare (Casasola et al., 2009).

Although the basic definition of biodiversity is simply “the diversity of life on Earth” (MA, 2005), assigning an economic value to biodiversity, especially at the farm level is an extremely difficult task. Biodiversity has both intrinsic and instrumental values. The latter is to some extent easier to estimate as it includes revenues from ecotourism, bioprospecting tasks and services that already have a market value. Intrinsic value on the other hand is based on existence value in time and space. Biodiversity is valuable for its separate components (e.g. food), but also because the diversity of species affects the capability of an ecosystem to be resilient to changes in the environment. Valuation of these services is limited by the poor knowledge and understanding of their biophysical parameters and the non-existence of markets for those services (Chomitz et al., 1999). The Convention on Biological Diversity (CBD COP VII/30) suggests that biodiversity can be lost either if the diversity per se is reduced (such as through the extinction of some species) or if the potential of the components of diversity to provide a particular service is diminished (such as through unsustainable harvest).

Measuring watershed services has received increasing attention, although market formation does not necessarily reflect scientific evidence (Muradian et al., 2010; Porras et al., 2008). The connection between land use, forest and water is complex and highly dependent on local conditions such as vegetation cover, soil characteristics, and weather (Alpizar et al., 2007; Bonell and Bruijnzeel, 2005; Calder, 2005). There is for example evidence on the benefits of forest cover on sedimentation (reduced) and peak flows. However, such benefits on total annual flows have not been proven in many cases, and there is evidence proving the contrary for reforestation (Blackman and Woodward, 2010; Calder, 2005), or pointing at the relatively modest impact of positive relationships, for example, of cloud forests (Bruijnzeel et al., 2010).

2.2 Environmental effectiveness

From a technical perspective, PES effectiveness is determined by leakage, additionality, permanence, targeting and social efficiency (Engel et al., 2008; Wunder, 2007).

Leakage or spillage occurs when activities that generate negative externalities are displaced to areas where the PES programme has not been implemented (Engel et al., 2008).

Additionality is how much more of the environmental service is available because of the introduction of the payment, and it is important in order to avoid making payments for nothing or paying for the adoption of practices that would have been adopted anyway (Engel et al., 2008). According to this, PES are effective if the payments result in the delivery of environmental services that would not have happened in the absence of payments (Pattanayak et al., 2010).

The use of the precautionary principle is sometimes used to justify projects, undermining the additionality rule. This principle suggests that under risk of deforestation a land can be protected to ensure provision of its environmental services until such a time when scientific information is produced (Asquith and Wunder, 2008).

Permanence refers to the ability of a PES programme to improve the provision of environmental services in the long run, even after payments have ceased. It depends on the programme's capability of adapting to changing conditions (i.e. market conditions, lack of funding) and therefore relies on the key feature of a PES regarding its voluntary nature and the ability of authors to renegotiate contracts (Engel et al., 2008). The likelihood that the benefits of PES programmes would continue once the programme expires depends on the permanency of the underlying PES externality (Wunder et al., 2008). Conditionality is a key component of permanence. It consists on ensuring the environmental services are actually provided. For it to be effective, a proper monitoring programme is needed as well as enforcement of sanctions for non-compliance (Pattanayak et al., 2010).

Targeting is needed when demand for funding (or supply of ES) exceeds available funding resources. It can be done based on benefits issues, cost factors or a combination of both. Targeting not only requires the selection of the appropriate location but also choosing the resources that will deliver services of the required quality (Pattanayak et al., 2010).

Social inefficiency (discussed in more depth in a separate section) in terms of enrolment arises when the programme fails to enrol practices whose benefits are higher than its costs or when practices whose costs are higher than their benefits are adopted (Engel et al., 2008; Pattanayak et al., 2010).

2.3 Cost-effectiveness

Cost-effectiveness is increasingly recognised as a key requirement for gaining social and political acceptance for costly conservation measures (Wätzold et al., 2010; Wätzold and Schwerdtner, 2005). A key feature of PES in conservation is that it does not require large upfront investments for buying land. However, there are other significant costs involved and they fall differently on the actors involved.

There are two main cost categories for providers of environmental costs: *transaction costs*, which include application and costs associated to permanence in the programme and, *compliance costs* that encompass the costs associated with the required changes in land management and practices to ensure service provision. Transaction costs represent an important share into the total costs of PES, and so far experience suggests that they are “sobering” (Wunder et al., 2008).

Administration costs are correlated to the type of activity subject to payment, and the number of participants and economies of scale (Grieg-Gran et al., 2006; Wunder, 2007). In general, programmes aiming at maintaining land use are less costly than those that focus on changing the economic activity performed such as reforesting land that has previously been deforested for agricultural purposes (Engel et al., 2008). Engel et al. (2008) suggest that the most efficient PES programmes are the ones where the buyers are the final users of the ES. In such a scheme, actors have the incentive to ensure and monitor the appropriate provision of the service; can renegotiate the agreement when needed and have the best information regarding the value of the service. On the other hand, the authors argue that third party buyers (not the ones enjoying the service) do not have first-hand information on the value of the service and they do not have the ability to secure the ES is being provided. User-financed programmes are likely to emerge when the ES is a private good, the PES benefits a small number of actors, incentives for free-riders and coordination costs are low and when the ES benefits are large enough for every actor to have a big enough share of them. Contrary to this, it would be hard to institute a user-financed programme when the ES provided are public and there are a large number of actors involved (as the incentive to free-ride increases). In such cases, government-financed programmes are the only option to establish a PES programme (Engel et al., 2008). However, government-financed programmes seem to limit minority group participation through land tenure requirements while user/private financed programmes allow for wider participation of the poor (Pattanayak et al., 2010).

Cost-effectiveness increases if payments reflect producer’s opportunity costs. For example, the United States distribute their payments from the Conservation Reserve Program (CRP) on the base of bids from the farmers. This auction-based programme limits the possibilities, in comparison to a fixed-rate payment, for the local farmers to earn excessive rents and ensures more environmental benefits per dollar spent (Baylis et al., 2008).

2.4 Reported impacts from experiences

2.4.1 Environmental effectiveness

In Europe, evidence on the effectiveness of PES schemes in terms of biodiversity improvements is limited. For example, the success of the forestry programmes in Austria, Finland and Sweden is measured in the number and area of forests enrolled in the programme (Mayer and Tikka, 2006), not in terms of biodiversity indicators. A review of agri-environment schemes in the EU (Kleijn and Sutherland, 2003) did not allow for a general judgment on their effectiveness because of a lack of sufficiently rigorous studies. Moreover, while most of the studies reviewed found increases in biodiversity, there were also decreases.

An example of how an appropriate definition of the baseline improves the likelihood of success of PES programmes is found in the GED/Silvopastoral project. The scheme was executed by CATIE in Colombia, Nicaragua and Costa Rica. The compensation framework was defined so that farmers received payments on the basis of annual improvements of their farm’s initial environmental index (baseline). The scheme was able to reduce degraded pastures in 14.2%, increase the use of improved pasture in 39.4% and forest area by 1% (Casasola et al., 2009).

Evidence suggests that Costa Rica’s and Mexico’s PES programmes had not reached the desired effectiveness levels due to lack or inappropriate targeting of the beneficiary lands and services provided (Barton et al., 2009; Muñoz-Piña et al., in press; Robalino et al., 2008; Sierra and Russman, 2006). In Costa

Rica, although hydrological and biodiversity benefits figure highly on the agenda, by 2005 only about one third of enrolled parcels were located within a basin with downstream users of hydrological services; and between 30% and 65% of parcels were key in biodiversity conservation (Blackman and Woodward, 2010). Barton et al. (2009) showed that compared to its 1999-2001 period Costa Rica's second PES phase achieved a higher cost-effectiveness relative to biodiversity surrogate indicators and opportunity cost to forestry and agriculture thanks to improved targeting of biological corridors and buffer areas. Evaluators of Mexico's programme highlight the problems of how targeting can clutter a programme: information shows that only one third of enrolled areas were under risk of significant deforestation, and increasingly criteria other than environmental ones diminish the programme's ability to be environmentally effective (Alix-Garcia et al., 2005; Muñoz-Piña et al., in press; Muñoz-Piña et al., 2008). Voluntary conservation contracts, where landowners determine the supply and spatial location of land (e.g. Norway and Finland) have been criticised because they may result in a patchwork of conserved locations which does not fulfil ecological functions needed to conserve biodiversity. Barton et al (2009) also show a clear trade-off between PES targeting of biodiversity representation versus landscape function. Analysis of the Trading in Natural Values in Finland shows that while the programme meets its ecological goals, it is still early to assess the long-term ecological effects (Juutinen et al., 2009).

Environmental effectiveness can also be affected by the project's ability to reach a threshold. Small schemes of a few hectares, unless strongly targeted, are less likely to make a significant impact on the delivery of the service. On the other hand, analysis of Chinese PES schemes, where most programmes are considerable big, show different success rates. In the Chinese Sloping Land Conversion Programme, nine million hectares of sloping land has been converted into forestland and tree plantations by 2005, with different rates of success in terms of tree survival. Important impacts have been reported in terms of sediments, with silt run-off from converted lands 22-24% less than from comparable farming lands in Tianquan County (Sun and Chen, 2006). Between 2000 and 2006, China has planted 28 million hectares of plantations in 6 years. The Conversion of Cropland to Forest Programme (CCFP), which pays farmers to plant trees rather than crops, has converted 8.8 million hectares of crop to plantations. Soil erosion has been reduced by 4.1 million ha. Desertification has been successfully tackled through the Sand Control and several Shelter Belt Programmes. The total plantation area in China amounts to 53 million hectares.

PES effectiveness can be affected by many and varied confounding factors, ranging from personal characteristics of the participant and their land, to economic variables that may affect deforestation risk (see Table 3).

Table 3. PES confounding factors in Costa Rica

Confounding factors	
Determinants of programme participation	Population age, education, local residents Distance to MINAE/SINAC regional office Existence of an intermediary
PES spatial targeting	Distance to agricultural settlements (IDA) Proportion located above aquifers Proportion located in Ecomarkets/KfW areas Located in conservation planning areas (GRUAS)
Immediate causes of deforestation	Land-use capacity (LUC: soil type, slope, elevation, water logging) Off-farm employment Distance to roads Distance to markets Scale (tract size and forest stock) Proportion households using fuel-wood
Underlying causes of deforestation	Population Proportion immigration

Source: Arriagada (2008)

2.4.2 Economic impacts

Economic impacts can be measured through several indicators such as cash flows, jobs generated/lost; land tenure as discussed below.

Programme financing issues

Perhaps the most important economic impact of PES schemes is the flow of money it brings. In most ongoing schemes of watershed environmental schemes (Porras et al., 2008), landowners (private or community) receive a payment (cash, in-kind, or both) that can be continuous or one-off. Regardless of the size of the payment, these transfers are important for the family budget, enabling participants to invest in, for example, land or home improvements, payment of debts, and access to medical services or community infrastructure in the case of communal payments (Tacconi et al., 2010). The degree of the impact depends, ultimately, on the size of the operation, on the household dependence on their land and their access to alternative (non-forest) income. In Pimampiro, Ecuador, although payments are low (US$ 1/ha/month), they are reported to reach up to 30% of average household income while payments in Nicaragua and Honduras are not able to compensate opportunity costs of non-conservation alternative economic alternatives (Porras et al., 2008). The Vittel scheme in France has average payments of € 200 per hectare/year over a five year transition period and up to € 150,000 per farm to cover costs of new equipment (ten Brink et al., 2011).

The value of carbon under REDD in forests in developing countries can contribute to significant cash flows. has been estimated to be at over US$ 43 billion if REDD projects are formalised. Forested areas could be worth US$ 200-10,000 per hectare depending upon a number of factors such as carbon content and project type (Peskett et al., 2008). Protected areas can derive a significant income from environmental services. A 2008 valuation of all of the carbon stored in ecosystems within protected areas estimated its worth at approximately € 5,700 billion (Campbell et al., 2008). A valuation study of ecosystem services in a national park in Costa Rica suggests that the Park could be potentially receiving US$ 43 per hectare per year, even without counting benefits in terms of water (Bernard et al., 2009). Similar opportunities exist for projects targeting carbon and biodiversity. Initiatives such as Biodiversity Banking acknowledges uncertainty in timber markets, and suggest that forest operators can widen up their approach to receive benefits such as carbon offsets, sales of biodiversity credits, public relations and investment risk spreading (Berkessy and Wintle, 2007; Blundell, 2006; ten Kate et al., 2004).

Jobs

In Costa Rica, reforestation promoted through the Payments for Environmental Services programme has had a modest impact on local employment, infrastructure and micro-enterprise development (Miranda et al., 2003; Tacconi et al., 2010). Increasing areas into conservation can take areas from other economic uses, having potential negative impacts on landless workers. In other cases, it is still early to assess whether the PES will lead to lasting improvements in poor participants' welfare, for example in the Silvopastoral project in Nicaragua (Pagiola et al., 2007), and an evaluation of PES in the Osa Peninsula in Costa Rica (Muñoz-Calvo, 2004). Changes in forestry policies have impacts outside the forestry sector. The Forest Protection Programme (NFPP) in the Southern provinces of China has led to an increase of off-farm labour supply more rapidly than compared to non-NFPP areas (Groom et al., 2008). Meanwhile, the impact on labour allocation to the most lucrative off-farm activities outside the villages in the Sloping Land Conversion Programme in China has been limited, mostly because of institutional constraints to labour movements (Groom et al., 2008).

Other economic benefits

Examples of in-kind payments are seeds provision, roads, school or health center building and grants to community-based organisations, access to other kind of royalties, and using other economic instruments such as tax credits (Bond and Mayers, 2010; Meijerink, 2007; Porras et al., 2008). PES can increase the value that local communities have for their forest resources when dealing with illegal logging interests; but ultimately their decision to sell-out depends on their capacity to enforce conservation in their lands, a crucial problem in situations of weak property rights (Engel and Palmer, 2008; Engel and Palmer, 2010).

Most ongoing PES schemes operate on the basis of some form of secure land tenure for participation. In places dealing with indigenous lands (like carbon forestry payments in Mexico and the 'no fire bonus' in the Philippines), the existence of indigenous resource rights and institutions is regarded as a key factor in the operation of community level payment schemes (Tacconi et al., 2010). In Costa Rica the PSA programme has had a moderate to limited impact on land tenure and security, as the vast majority of participants have property rights (Porras, 2010). However, in places where there is a lack of formal government-approved property rights (e.g. Bolivia) some investors recognise the necessity to work with de-facto property rights recognised by local farmer unions (Asquith and Vargas, 2007).

PES schemes frequently strengthened resource management and social coordination capacities of the community institutions they worked with (Tacconi et al., 2010). Capacity building is commonly reported as a benefit from PES schemes, for example, increasing agriculture productivity in Pimampiro, Ecuador (Echavarria et al., 2003), apicultural training in Bolivia measured at US$ 35 per participant (Asquith and Vargas, 2007). However, for Tacconi et al. (2009) there is little evidence available about the long-term impact of capacity building activities, for instance whether new knowledge and skills were applied in practice. Programmes like PES contribute to strengthen the capacity of existing forest institutions, and their relationship with landowners.

Non-tangible benefits

These benefits are also important drivers of participation. A study on drivers of participation in watershed projects in Costa Rica shows that firms entering contracts consider possible non-tangible or measurable benefits they could achieve through participation such as changes in people's perception about the firm, better relationships with the community and political allowances (Blackman and Woodward, 2010). This finding supports the conceptual ideas expressed by many authors in the sense that compensations can be either in cash or in-kind.

2.4.3 Equity and social justice

There are trade-offs between environmental and social impacts of PES projects (Grieg-Gran et al., 2006). Some authors consider this dichotomy as a drawback of PES programmes since worse-off landowners should be able to participate in PES programmes regardless of the impact their participation could have on its effectiveness (Muradian et al., 2010). In this chapter we will avoid such discussion and will focus on discussing evidence of the impacts of existing PES both at the environmental and social levels.

Although there are some who consider that poverty alleviation should not be a direct goal of PES programmes, concentrating only on economic and environmental issues can potentially lead to three problems (Adger et al., 2002):

I. It creates a narrow understanding that trivialises actual difficulties in making choices and in implementing them through the required institutions;

II. It results in promotion of partial solutions, which tend to be rendered illegitimate by those outside that perspective; and

III. Final decisions can result in unexpected consequences and fail to realise sought after goals.

A very recent example on the potential of avoided deforestation in Brazil suggests that at current carbon prices the economic preconditions exist for over half of threatened forests over the next decade. However, important institutional conditions, like land grabbing, insecure tenure, overlapping claims, and lacking information on private tenure, are serious impediments to PES. Inequity implications are large. If nothing is done to address current tenure insecurities, large landowners (who account for about 80% of all deforestation, have lower opportunity costs, and are more likely to deliver higher cost-effectiveness to the scheme) would be the highest beneficiaries (Börner et al., 2010).

Equity in PES schemes can be assessed using a three-tiered framework (Brown and Corbera, 2003):

- Equity on access: which individual farmers, rural communities and organisations are able to participate in emerging markets. Related to access to information, knowledge and networks, land and forest resources.
- Equity on decision-making: procedural fairness within the project framework. Relates to issues of recognition and inclusion in strategic management decisions (Paavola, 2003).
- Equity on outcome: distribution of project outcomes across participants (and non-participants) of economic payments (and other benefits) and their perceived fairness (also linked to legitimacy). Distribution of project outcomes will in turn be determined by access to project activities and decision-making (e.g., those without a voice in project management may not be able to benefit from specific outcomes, such as forest management training activities).

Equity of outcomes

The social nature of PES programmes comes from the increasing interest in combining economic growth, environmental protection and poverty reduction goals in the definition of market based mechanisms to better manage ecosystem services (Meijerink, 2007). However evidence suggests that PES programmes have not been successful in poverty alleviation or in the best of cases have had a mixed effect. PES in particular has a potential to benefit, and harm, poorer households (Grieg-Gran et al., 2005; Pagiola et al., 2005). In theory, poorer actors, those located in areas with less economic options, and many indigenous groups are likely to become ES providers because of economic (relatively low opportunity cost) and geographic reasons (assuming that their lands are located in marginal sites, such as steep slopes, poor soil, where the ES are more under threat). In practice, the opposite is more likely to happen, with wealthier landowners in possession of more and better assets, with access to livelihood options which do not depend on the land, larger forest areas they can enter into protection, better connected and informed, and just as likely (or more) to receive payments if they happen to live in the target areas (Landell-Mills and Porras, 2002; Hope et al., 2005; Pagiola et al., 2005; Pattanayak et al., 2010; Porras, 2010; Zbinden and Lee, 2005).

Theoretical work shows that several variables, most independent of the environment and the programme definition, determine the social impact of PES projects (Zilberman et al., 2008). For example, the authors conclude that if food demand possesses a high elasticity, urban and rural poor can be harmed by PES programmes if they are to affect food supply. Similarly, countries or areas where labour markets are not integrated PES projects might make rural poor worse-off as labour demand decreases. One of the most important messages from the work of Zilberman et al. (2008) is that social evaluation of PES programmes must not only incorporate income earnings but also the impact of such programmes on the cost of living.

Access

Analysis of the Costa Rican experience identify a positive relationship between the likelihood of participating in PES and farm size, human capital and economic factors and access to information (Zbinden and Lee, 2005). This implies that small farmers and landowners have been left behind in the programme design. If having a positive social impact for poorer actors were an explicit goal of the programme, authorities must start working on targeting those populations.

At the same time, poor landowners do not have the resources needed to cover transaction costs associated to PES application and possible implementation. As Engel et al. (2008) argue, the higher the transaction costs the less participation of low income actors. Keeping transaction costs at their lowest possible level will ensure poor farmers participation (Grieg-Gran et al., 2006). Pagiola (2005) (see Figure 2) suggests that the obstacles faced by poor farmers to participate in PES are:

- **Spatial eligibility** (i.e. applicants located in areas that provide environmental services).
- **Property eligibility:** linked to landless people; property or possession rights; minimum and maximum property sizes; existence of forest cover (i.e. excludes subsistence agriculture); transaction costs.
- **Desire to participate:** linked to expected profitability, individual's opportunity cost (linked to farm and off-farm activities; how it fits in the overall farm system; Includes direct payments, contribution to cash income, alternative activities. Benefits at timescale. Discount rates issues. Opportunity costs. Stability and continuity of payments. Payments uniform or differentiated. Non-monetary benefits).
- **Ability to participate:** looks into tenure, investment costs, access to savings/off-farm income; credit; collaterals, payment schedule; technical constraints; and transaction costs: a) Programme manager costs of running the programme, and b) the costs imposed on participants themselves. Issues include economies of scale, pairing down requirements; collective contracts.

Even those small and poor farmers who enter the programme can experience reduction in their income from unexpected costs for reforestation, or lack of capacity on how to secure greater income from their forests (Locatelli et al., 2008). In the Mexican PES programme, the poorest farmers and women have been excluded from project design and implementation (Brown et al., 2004). Pitfalls such as these in emerging markets contribute to reinforcing existing power structures, inequities and vulnerabilities. So far, markets for ecosystem services are, in effect, limited in promoting more legitimate forms of decision making and a more equitable distribution of their outcomes.

On the other hand, voluntary market approaches like PES can help increase legitimacy of conservation measures. Traditional compulsory conservation is usually linked to evictions and violation of landowners' rights (either legal or de-facto). In the Natura 2000 implementation in Finland, this kind of lack of legitimacy of the enforced conservation led to escalated conflicts (Hiedenpää, 2005). Compared to these conflicts, the landowner rights and legitimacy of policy experienced by landowners are much stronger in the voluntary conservation contracts, which also acknowledge the utility and beliefs that landowners place on biodiversity (Paloniemi and Tikka, 2008; Paloniemi and Varho, 2009). A similar experience of voluntary conservation has evolved in Norway over the past decade. It remains unclear whether a voluntary approach will delivery enough of the forest types currently under-represented in the public protected areas to meet conservation targets (Framstad and Blindheim, 2010).

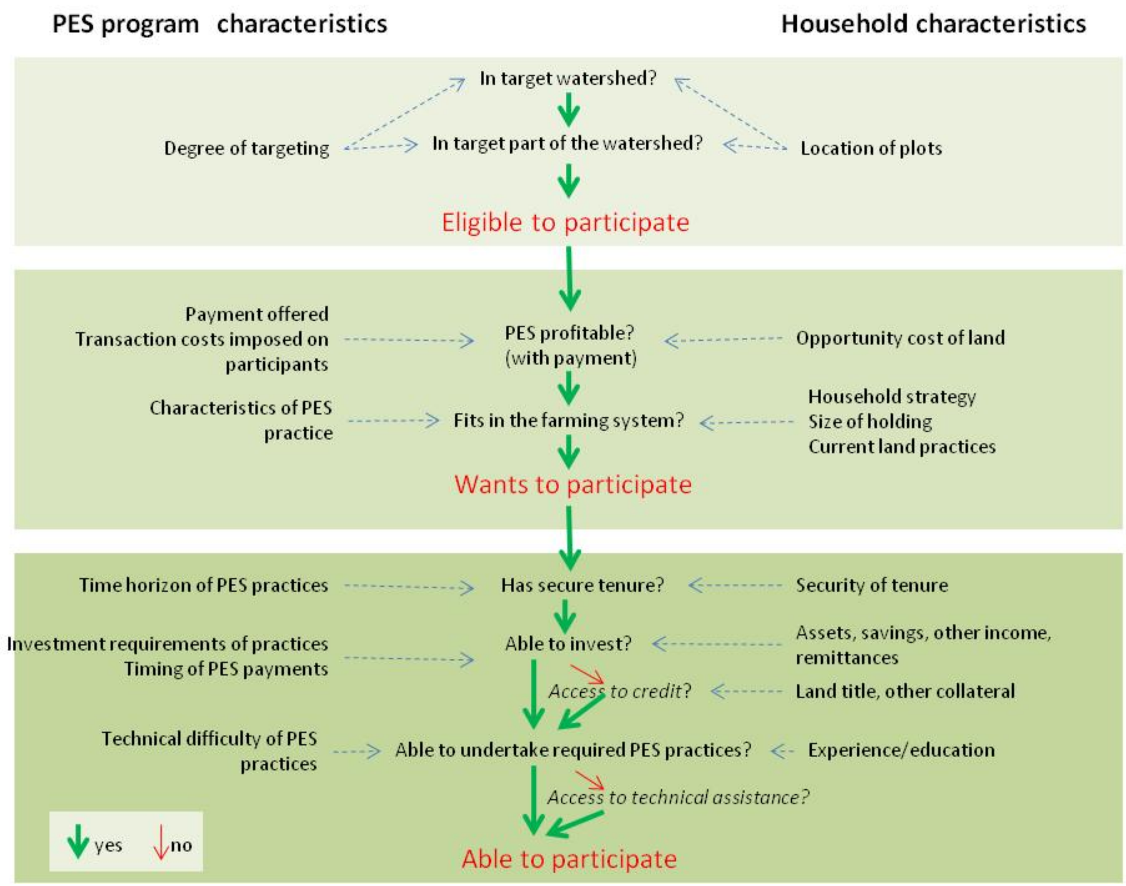

Figure 2. Factors that affect household participation in PES programmes

Source: Pagiola et al. (2005)

2.5 Institutional context and requirements

Governance consists on establishing goals, defining the rules to achieve those goals and the control of outcomes once goals have been reached (Vatn, 2010). In a PES context, governance can be summarised as the definition of the environmental service to be compensated, demarcation of the way those resources will be managed to ensure the desired provision of the service and of the rules of conduct for the actors involved and finally, how outcomes are to be monitored and used. In other words, it refers to institutions at all levels that are required to ensure proper functioning of PES programmes. From a practical perspective, PES success depends on institutional engagement either at the national or community governance level (Vatn, 2010). However for some, civil society institutions and NGO´s are the most effective way of increasing PES's efficiency and equity (Grieg-Gran et al., 2006), which diminishes the weight or importance national institutions have on PES functioning.

Nonetheless, national institutions (or governance) are required as they are the ones that frame the application of the programmes and the establishment of the PES setup. Existing governance structures, rules and property rights, influence the way landowners or other providers of ecosystem services react to the establishment of a PES scheme. The providers of services base their decisions on historically molded traditions, norms and motivations, as well as on the sources of knowledge and beliefs about the service, and about the interactions between the governance system, the service, and its provision or use (Ostrom, 2007; Vatn, 2010). For some authors, the goal of institutional arrangements around PES programmes should be to reduce transaction costs and to maximise the benefits to ES providers. In this sense, institutions should respond to the different ES provided (Eaton and Meijerink, 2006).

Another purpose of national governance for PES is the definition of property rights are a key factor in the success of PES programmes. Private property rights are not necessary as long as there is proper definition of resource ownership, and it is possible to develop systems that "*propertise*" ecosystems and their services with not need privatising them (Farley and Costanza, 2010). These authors argue that institutional arrangements should adapt themselves to existent property rights as much as possible to democratise PES participation, for example in communal lands in Mexico (the Ejidos) and Brazil (municipalities). If privatisation is necessary, institutions should adapt to the specific problem; they can take for instance the form of public sector organisations, international protocols, conventions, or treaties, or a commons sector.

In general, and independently of their main goals, institutions must be trusted. For instance, the success of Finland's voluntary forest biodiversity conservation contracts is thought to have been granted by the autonomy given to landowners and the confidence those actors had on the organisations in charge of communicating the programme (Paloniemi and Varho, 2009; Paloniemi and Tikka, 2008). Similarly, voluntary forest conservation contracts in Norway have been responsible for the majority of new forest areas under either public or private conservation, indicating mistrust landowners previously felt to publicly initiated conservation.

3 The role of the instrument in a policy mix

PES is designed to complement existing legislations regarding the use of ecosystems (i.e. cap and trade), and to help align local malpractices through negotiation between parties where no legislation exists. In practice, PES should not work in a vacuum, and it is not a 'fit for all' measure (Echavarria et al., 2003; Engel et al., 2008).

Before even considering PES as a potential instrument, it is important to understand the nature of existing market failure. For example, whether it is from lack of ability to manage and enforce decisions over land, lack of information about potential private gains from improved land management, or a gap in capital markets preventing farmers from adapting privately profitable technologies or practices that enhance ecosystem service provision. A review of existing cases shows that implementation of PES has more chances of succeeding if:

- Landowners recognise that ecosystems are mismanaged because many of their benefits are externalities. Allocating property rights only, or raising environmental awareness, may still not be sufficient to make conservation economically attractive as alternative land uses (Engel et al., 2008; Palmer and Engel, 2007);
- Local managers have the authority to make decisions over land. This means that either there are property rights, legal or de-facto, which means that someone has the ability to enforce a land use over the duration of the contract;
- Transaction costs are low or manageable, compared to other options;
- Service users are easy to identify, approach, and have ability to pay;
- Free riding can be kept under control, so that the externality is sufficiently large to generate an incentive for the land manager to change.

Table 4 presents a summary of the main issues related to PES and other command-and-control measures. In Europe, voluntary payment schemes for watershed services are uncommon, since the rights and duties of the actors involved are generally fixed by regulations[3].

In addition, many instruments overlap with each other. For example, PES are considered a type of subsidy when compensations are made directly to the land managers. However, payments from service users can come as a result of an environmental tax, or extra user-fees (for example, added to water charges).

PES can be used to 'sweeten' prohibitions on the use of forests as funds for such programmes from users are easier to obtain if legislation exists on the nature of ecosystem services.

4 Concluding remarks

PES are effective in providing extra compensation required to cover costs of forest protection and management, and improved agriculture practices that will help deliver better ecosystem services. Conditionality is the main characteristic that separates PES from other instruments, and adds weight to its potential applicability in international trade. Although early schemes have taken conditionality in a relaxed way, it is bound to be more important as the tool hits the market under REDD schemes, or as more local users are incorporated and demand value for money. International experience is not evenly distributed. In Latin America, PES is a known tool that is being used. In Southeast Asia, it is a tool that is starting to be developed while in Africa there are very few examples of PES projects and programmes, although this keeps changing rapidly. In Europe, the demand side of PES is usually represented by the government, the main example being agri-environment schemes co-financed by the EU's Common Agricultural Policy.

[3] *For example, the EU Water Framework Directive (WFD) requires Member States to prepare River Basin Management Plans with a view to achieve good water quality by 2015. The WFD also stipulates that the principles of cost recovery and 'polluter pays' should be applied to all water services.*

Cost-effectiveness: Overall, little is known about cost-effectiveness of existing PES schemes, and even less how they compare to other policy instruments. Evaluation studies are scarce; there is a tangible absence of baselines, difficulty to control for confounding factors and difference-in-differences, and strong monitoring to inform project design has not remained a priority. In theory PES should be sleek creatures, where service providers make offers for the contracts based on their own private assessment of opportunity costs. Buyers make payments based on the value of the environmental service, the perceived level of the threat that the ecosystem will change, the perception of how this will affect their reputation, a compulsory fee imposed by a third party (i.e. municipality or national government), or a combination of all of the above. Cost-effectiveness of PES is evaluated based on those opportunity costs, the costs of implementing changes when they are required, and the transaction costs of the programme. Costs are correlated to the type of activity subject to payment (i.e. if expensive changes are required), and the possibilities of economies of scale (as opposed to fragmented, small parcels). Start-up costs are very high, and they can include setting up the scheme, baselines, contract negotiation, fundraising, and awareness campaigns. Cutting corners is a common practice, for example, few public consultations and badly done baseline studies, although it may have negative effects in the long-term regarding project uptake.

The level of the incentive needs to be addressed, and payments need to reflect the type and level of benefits they provide. A discrepancy between practice and theory is who pays. Ideally, beneficiaries are the ones that should pay providers based on the opportunity costs and knowledge of both agents. In reality, most PES programmes are financed and driven by governments, donors or other outside institutions, with little negotiation between providers and users. Ensuring long-term finance is one of the main challenges for many schemes.

Impacts on the poor: Evidence suggests that PES programmes have not been successful in poverty alleviation or in the best of cases have had a mixed effect. PES in particular has a potential to benefit, and harm, poorer households. In the practice, PES tend to benefit a large proportion of wealthier landowners in possession of more and better assets, with access to livelihood options which do not depend on the land, larger properties, better connected and informed, and just as likely (or more) to receive payments if they happen to live in designated social target areas. Transaction costs tend to be fixed for the provider, and the higher the cost the less likely poorer household will enter, and in many developing countries the poorest farmers, indigenous groups without connections, and women have been excluded from project design and implementation. Pitfalls such as these in emerging markets contribute to reinforcing existing power structures, inequities and vulnerabilities. So far, markets for ecosystem services are, in effect, limited in promoting more legitimate forms of decision making and a more equitable distribution of their outcomes in the developing world context. The situation may be different in developed countries (Norway and Finland), where the introduction of voluntary contracts seems to increase legitimacy and sense of justice, as opposed to compulsory conservation.

Governance levels: Even if financial issues are resolved, institutional and distributional concerns can affect the performance of the instrument. The inclusion of multiple objectives (environmental and social, especially in national programmes) tends to result in less sophisticated instruments. Local schemes, and those financed privately, tend to be better targeted and monitored, are more likely to improve delivery of environmental services, are more attuned to local conditions and necessities, and have a greater willingness to implement conditionality.

Table 4. The role of PES in a policy mix

Instrument	Economic issues	Equity issues	Political and institutional issues	Overlap
PES	The subsidy nature of PES implies a level of potential inefficiency, in the form of lack of additionality and sources of leakage, and the possibility of becoming a perverse incentive. By artificially raising the profitability of an activity it can lead to inefficient allocation of resources, and can also be used for protectionist purposes.	In developing countries farmers are considered less well-off than service users. Little attention to equity, procedural and distributional justice, and legitimacy of the instrument).	Powerful agriculture groups in developed countries push for subsidies rather than taxes (Engel et al., 2008).	
Taxes	Directly address the nature of the market inefficiency. Problematic when incomplete markets.	Imposes the cost of conservation on landowner.	Not politically defendable in many places.	Taxes and user fees are commonly used to collect money and pay land managers for externalities.
Command-and-control (prohibitions, national parks, etc.)	Less efficient than PES as it tends to prescribe the same level of activity to land managers, while market-based instruments are deemed more flexible.	Imposing restrictions on forests can have significant negative impacts for local groups whose livelihoods depend on these ecosystems.	In developing countries it is restricted by weak governance, high transaction costs and information problems.	PES coexist in many places with command-and-control: Makes prohibitions 'more palatable', for example those evicted from reserves, people living inside or in buffer areas of national parks and reserves who have restricted activities; Weakly enforced regulations can reduce the expected gain from non-compliance. By raising the value of the ES, it can increase local people incentives to self-enforce restrictions, overcoming the need for government regulation. Command-and-control provided property rights necessary to secure rights to exclusion and increase reliability of providers.
ICDP (Integrated Conservation Development Project)	Success rate low (Ferraro and Simpson, 2002). No conditionality. One-off, up-front payments make it hard to pursue compliance.	Strong social component: projects aim at providing communities with alternative livelihood options. However, these options are not always new and not additional.		

Sources: Börner et al. (2010); Landell-Mills and Porras (2002); Engel et al. (2008)

Their role in a policy mix: PES is designed to complement existing legislations regarding the use of ecosystems (i.e. cap and trade), and to help align local malpractices through negotiation between parties where no legislation exists. PES coexists in many places with command-and-control, making prohibitions 'more palatable', for example those evicted from reserves, people living inside or in buffer areas of national parks and reserves who have restricted activities, and increasing self-enforced restrictions by raising the value of the environmental services. By focusing on variable payments, PES has more chances of success than Integrated Conservation Development Projects (ICDPs) and the lessons from attaching social objectives to environmental policies used in ICDPs are valuable material for PES schemes.

Trade-offs are highly likely to occur, whichever instrument is used. However, inconsistencies between practice and theory are responsible for the lack of success PES seems to have had in protecting ecosystems and reducing poverty. Some command-and-control instruments (such as protected areas and/or legislation) must be simultaneously addressed. On the other hand, if poverty reduction is the main goal of the programme, authorities must address institutional poverty factors to accompany the project (such as improvements in health, education, and sanitation), which a PES on its own will not be able to address.

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Reduced Emissions due to Reduced Deforestation and Forest Degradation (REDD and REDD+)

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Summary

Reduced Emissions due to Reduced Deforestation and Forest Degradation (REDD) is an innovative instrument that entitles developing countries to preserve forests that otherwise would have been cleared. It is justified on the fact that deforestation and forest degradation constitute the main sources of global carbon emissions, even exceeding gas emissions from the transportation sector. However, contrary to most market instruments, REDD has not been formally implemented yet and its key features (and to some extent its definition) are still being negotiated. Under this scenario we discuss important issues related to its expected effectiveness, cost-efficiency and social impacts and try to address its role in a wider policy mix for conservation.

1 Definition and key features

The proposed mechanism known as "reduced emissions from deforestation and forest degradation" (REDD) is a voluntary instrument that entitles development countries to reduce emissions from deforestation and forest degradation (Bosetti et al., 2008). Under this framework, REDD entitles monetary compensation to individuals, communities and countries for their reductions in carbon emissions from forest degradation and deforestation which funds are to come from industrialised countries (Angelsen, 2008). The approach is justified on the need of addressing deforestation and forest degradation as they constitute the main source of global carbon emissions, even exceeding gas emissions from the entire transportation sector (Angelsen et al., 2009).

The concept was formally included in the international climate negotiations during the COP 11 of the United Nations Framework Convention on Climate Change in Montreal in 2005 (Schwartzmann et al., 2008; Sikor et al., 2010; UNFCCC, 2005a) in response to a joint proposal made by Papua New Guinea and Costa Rica, with the support of eight other Parties. They proposed the inclusion in the agenda of an item on "Reducing emissions from deforestation in developing countries: approaches to stimulate action" (UNFCCC, 2005b).

After the COP acceptance of the instrument, a consultation processes was called focused on the scientific, technical and methodological issues and exchange of relevant information and experiences. Later on, in 2007 REDD was included in the Bali Action Plan and it is expected to become a pivotal component of the post 2012 climate regime. A newer extension, discussed at the international level is known as REDD+, which interpretation varies along analysis, but in general it is expected to entitle compensation for activities related to forests conservation and sustainable management (Ring et al., 2010). An added benefit of the inclusion of conservation, sustainable management of forests and enhancement of forest carbon stocks in any possible future REDD mechanism allowed the negotiations process to move forward.

It is important to recognise that the mechanism is still being negotiated and therefore, its definition and key features have nor being finally defined. Discussions are currently based on many aspects such as the development of a multinational and multilevel payment for environmental services (PES) scheme and the preparation process tropical-forest countries must adopt to be prepared for its implementation (Angelsen, 2008); baselines; monitoring, reporting and verification (MRV); additionality: scope of REDD+, extent of integration with the carbon market; national vs subnational approaches; permanence; and, benefit distribution among others. The discussion presented in this chapter is therefore a discussion of recent developments and of what is expected once the mechanism will have been implemented.

Sometimes REDD is referred to as a specific policy directed at reducing carbon emissions (REDD) and biodiversity conservation (REDD+), but in its most general connotation rather than a single mechanism, it refers to a set of actions and policies involving actors at various levels: communities, countries, international (Angelsen and Wertz-Kanounnikoff, 2008; Boucher, 2008). The deforestation component refers to entire loss of forests through clearing and conversion to other land uses while degradation implies biomass loss in forests as result of harvesting forest products (Parker et al., 2009).

The PES[1] associated to REDD is said to be multinational as it must enable transference of financial resources from developed nations to tropical-forest countries, which are the ones that will ultimately implement actions to reduce deforestation and forest degradation (DD). The multilevel feature comes from the fact that the beneficiary countries must also foster a payment mechanism to compensate land users and communities for their efforts in reducing emissions as commanded by REDD (Angelsen and Wertz-Kanounnikoff, 2008).

A peculiarity that differentiates REDD from a *regular* payment for environmental services (PES) programme is that the former requires the definition of a baseline that would be used to evaluate the effectiveness in achieving the goals of the programme. In addition, credits to be generated by REDD actions are to be based on monitoring, reporting, and verification (MRV) and as such, a viable and efficient emission-carbon-measuring mechanism must be developed.

REDD requires the development of a nested governance structure involving international, national and sub-national institutional arrangements. At the most global scale is the international governance level, which is in charge of defining global rights; participation rules; and ensuring funding resources are to be based on firm commitments, are verifiable and enforceable (Sikor et al., 2010).

International arrangements are followed by a national governance organisation in forest countries that will be in charge of the definition of specific legal relations and procedures to be applied countrywide. Local governance, could be in charge of concrete rights and obligations regarding explicit resources and their distribution among individuals and stakeholders (Sikor et al., 2010), in coordination with national governance. Some particularities regarding each governance level are exposed below. The three approaches have implications for country programmes in terms of effectiveness, efficiency and equity as will be discussed in section 2.

The success of REDD will ultimately depend on the capacity of the host countries to implement their programmes properly. As such, country capacities must be developed in parallel to international institutions and capacities (Myers-Madeira, 2008).

[1] *For some, REDD does not necessarily recognise the existence of an international PES programme. However, since it entitles international transfers from industrialised to developing countries, one can think of the mechanism as a global PES structure.*

At the national scale, levels of implementation and national governance are directly related to geographical scaling for carbon reduction accounting and international crediting. Three possible scale levels are often discussed in literature: national, sub-national and a nested approach. The three approaches call for the development of national policies as well as international involvement. The main difference between the three levels settles on their national policy structure and their impact implications.

National programmes require the definition of country-wide policy reforms for carbon emission reductions from forest (Angelsen, 2008). Although it will probably succeed in bringing deforestation and forest degradation further down than the other approaches, a country's capacity to implement national plans is often constrained by institutional organisation and development.

Under restrictive conditions, a country's limited capacity to launch a national programme could depress participation of sub-national bodies that have the ability to join REDD programmes (Myers-Madeira, 2009). Sub-national programmes might be a good option in such cases. Smaller scale projects (sub-national) call for higher levels of private participation, which is incentivised by the fact that outputs are likely to be more tangible and present more profitable opportunities for private entrepreneurs (Angelsen, 2008). As in the case of the a national project, a sub-national approach will need governance and institutional development at the national scale to ensure that projects are properly functioning and the monitoring activities[2] are clear and according to global mandates.

Finally, the nested approach in a possible starting point for a country with no-national REDD programme to scale up from an initial sub-national approach to a national one. In addition, it opens to door for countries to receive carbon credits for both national and sub-national projects. Since it is envisioned as a more flexible option than only national projects, it allows for more countries to join REDD (Angelsen, 2008). National institutional arrangements are also needed to guarantee proper implementation and monitoring of REDD projects.

1.1 Actors involved

Among the many challenges faced by REDD and REDD+ mechanisms is the definition of benefits and responsibilities among participants at all levels. One of the most discussed issues focus on delimiting and safeguarding participation of local communities (The Nature Conservancy et al., 2010). However, there must be also good definition of the scope of participation at more aggregated levels, such as communities, regions and countries as a whole.

At the government level, REDD requires a deep knowledge of main deforestation triggers in each specific country and as such, REDD research provides a menu of SES context variables relevant for POLICYMIX case studies definition of institutional context. Previous forestry projects have shown that resources can go to waste if implemented without previous knowledge of the causes of deforestation (Chomitz et al., 2007). Since factors known to trigger deforestation rates in some countries are known to not have an impact on other countries deforestation (Myers-Madeira, 2008), a good starting point for government (public) involvement is to study the causes of deforestation in their respective countries. This does not mean we cannot learn from other's experiences but special attention must be given to differences among countries such as size, market development, institutional architecture, etc.

[2] *Section 1.2.4 describes in detail REDD monitoring activities.*

Governments must also take responsibility for financing (or looking for found sources) capacity-building and implementation cost needed to prepare their countries for REDD programme. Capacity building is related to technology, human capital, and material and institutional infrastructure. For example acquisition of satellite and remote sensing technologies, training of individual on issues related to REDD, building physical structure to monitor emission reductions, formalisation of land tenure, and enhancement of law enforcement capacity. Implementation costs are closely related and refer for example to forest monitoring, land tenure reforms, law enforcement, restrictions on road-building and land-use zoning (Dutschke et al., 2008; Boucher, 2008).

Participation of private actors will depend on the category of the country they are located. If in forest-countries, their main roll will be as executers of sub-national projects. If on the other hand, private actors are sited in developed countries in need of carbon emission off-setting, their primarily role will be financing carbon credits.

1.2 Baseline

One of the most challenging issues being debated in REDD, is the setting of reference levels or baseline that will serve as a base to calculate carbon emission changes and REDD effectiveness (Angelsen, 2008; Verchot and Petkova, 2010). In addition, defining a baseline entails the selection of a year of departure for measuring the effectiveness of the programme. This in turn is a strategic and sensible issue in the REDD negotiations process. Ideally in order to adequately define a baseline it would be required to have two identical countries, one where REDD have been implemented and the other without REDD policies. REDD effectiveness would be measured as the difference between deforestation and degradation rates. Pagiola et al. (2002) widely discussed this issue for the case of payments for environmental services. However, it is impossible to obtain two identical countries for comparison; think for example in the case of Brazil, it would be even difficult to find a similar country. So the question would be how we can define a final methodology to define a baseline. One of the problems is fitting a methodology to be applied to all countries since deforestation patterns and availability of forest inventories vary among countries (Myers-Madeira, 2008).

Some authors differentiate between the crediting baseline used to reward the countries from the benchmark to be used to measure effectiveness of REDD. Effectiveness should be based on a business as usual scenario (BAU) but there is debate on the method to be used to define the crediting baseline. The main argument to make such a distinction is that setting crediting reference levels tighter than the business as usual (BAU) scenario might prevent flooding the carbon markets crowding-out other carbon emission reducing initiatives (Angelsen, 2008).

In this chapter we will not make the differentiation explicit and will focus on the discussion of the main approaches suggested to define crediting baselines. In this sense, three main options are being discussed: historical deforestation, national circumstances and historical global deforestation. In this section, main benefits and drawbacks are discussed for each of those options.

Historical deforestation. In short, the proposal is to set the baseline for carbon credits based on each country's historical deforestation. Usually a 10 year average is suggested with updates made every 3 years. Although this is the most preferred baseline approach, many possible complications arise from it. The most obvious one is lack of data and how trustable available statistics are. Allowing countries some degree of flexibility on the definition of the years to be used would incentivise them to prefer data that would maximise the amount of credits they would receive.

National circumstances. This proposal aims at a baseline definition based on national conditions. Several alternatives have been made such as the inclusion of a development adjustment factor (DAF) to account for development differences; the use of elaborated models to predict future deforestation; and, categorisation of countries based in their deforestation rates.

The general DAF method proposes different levels of flexibility in the baseline based GDP resulting on low income countries getting more generous reference levels. The approach is justified by several arguments: (i) poorer countries are more likely to be on earlier stages of deforestation and therefore are subject to high deforestation rates under a BAU scenario; (ii) an inverse relationship between GDP and implementation REDD capacity is to be expected, needing relatively larger transfers; (iii) it would be expected that low income countries to face lower requirements under the UNFCCC principle of "common but differentiated responsibilities"; and, (iv) REDD should enable transfer to very poor countries (Angelsen, 2008).

An alternative approach is to define deforestation models based on national conditions such as population, forest area, economic growth, commodity prices, governance and location among others. Besides the limitations imposed by lack of data availability in many countries, reference levels defined under this method are dependent on forecasted variables (population, GDP, etc.), which might carry important biases (Angelsen, 2008).

An alternative proposal is to categorise countries based on their deforestation rates. A simple approach would be to have two groups: (a) one comprising nations with high deforestation rates, and (b) another including low-deforestation countries. Credits would be paid to type-a countries based on their deforestation reductions and to type-b ones conditional to them maintaining their historical low deforestation rates (Mollicone et al., 2007).

The problem arises with the definition of what are high and low deforestation rates. Some suggest using ***historical global deforestation***. Countries can be categorised into two groups depending on whether their deforestation rates are higher or lower than the global average. High deforesting countries would be those whose rates are higher than a half or a third of global average and, low deforesting countries would be those with rates lower than the threshold (Strassburg et al., 2010). Two implicit assumptions made by this method are strongly questioned. First, it assumes that deforestation is mainly consequence of "bad" conservation policies and not dependent on other factors such as economic growth, economic development and forest scarcity as has empirical evidence shows. Second, it presumes convergence of deforestation rates among countries in the long run, which is not supported by empirical evidence (Angelsen, 2008).

1.3 Monitoring, reporting and verification (MRV)

Given that carbon credits are to be translated into monetary compensation, monitoring is a key component of all REDD schemes. This was clearly stated in the 2008 report of the UNFCCC COP13 held in Bali in 2007 by indicating that "estimates of reductions or increases of emissions should be result-based, demonstrable, transparent and verifiable, and estimated consistently over time" (UNFCCC, 2008: 11).

The accounting system must contain at least four main features: (i) a forest inventory assessment or reference emission levels (baseline); (ii) a monitoring programme able to quantify forest changes; (iii) a common method to convert forest changes into carbon emission measures; and, (iv) a strong and impartial verification method (Myers-Madeira, 2009; Verchot and Petkova, 2010).

Monitoring and effectiveness of REDD projects and policies are closely related since the former will allow to gauge the real reduction attained under REDD. In fact, the decisions made in the Bali Action Plan established that demonstration activities must not only describe each action taken to reduce deforestation and forest degradation but also their degree of effectiveness (UNFCCC, 2008).

MRV activities are also to be effected by the implementation level chosen by the government: sub-national, national or nested (Angelsen, 2008). It would be expected that countries with low national monitoring capacity will favour sub-national approaches so monitoring activities are less demanding. This is under the assumption that subnational monitoring capacity will be more effective. Later on, monitoring capacities are to be developed based on past experiences enabling those countries to scale up their REDD projects up to an ideal national model.

Measures to incentivise countries to enhance their monitoring capacity are necessary. It has been proposed for example to implement a discounted credit approach where countries will get higher credits as they enhance their monitoring capacity up to a level where they qualify to receive full payments or credits (Angelsen, 2008).

There are also discussions on how REDD effectiveness is to be affected by the monitoring system. Some argue that it should be based on an output-based scheme in which credits are calculated based on actual emissions reduced. This scheme would also increase private funding opportunities and therefore participation as emission reductions and economic benefits will be more tangible. On the contrary, a stock-based mechanism could hinder effectiveness as it might imply big payments for non-under threat forest areas (Angelsen, 2008).

In addition, debate on credit center on whether allowances are to be based on gross or net emissions. The first option entitles payments based on vegetation replacement while the net approach requires accounting for actual carbon emission avoided. Gross accounting is less precise and would imply overestimation of REDD activities impact but is simpler to apply. On the other hand, monitoring based on net carbon sequestration and emissions is more precise although it is also harder to implement (Verchot and Petkova, 2010) and probably more costly.

REDD carries the additionally test, which results in a heavy burden for some countries and an advantage for others as early actions (previous to REDD implementation) might not be considered as additional in countries that tried to avoid deforestation in the past and have a relatively low baseline. This might not be the case for REDD+ actions (depending on the interpretation given), as its potential focus on biodiversity conservation might open the door to countries with a proven history of conservation policies.

2 Instruments performance

Since REDD is still a mechanism that is under definition, there are not actual evaluations of its performance. This section discusses elements that are considered would affect the instrument's performance but how successful it would be depends on its final definition.

2.1 Environmental impacts

2.1.1 Effectiveness

Effectiveness refers to the amount of carbon emissions reduced. From a global perspective, it will depend on the design of the REDD programme, country's commitment and its political feasibility. Effectiveness is also dependent on *additionality* (which requires reference level to be set at the business as usual scenario); *permanence* of achieved reductions (no temporal leakage); and, avoidance of geographical *leakage* (Angelsen, 2008).

REDD faces serious technical challenges with respect to permanence of climate (REDD) and biodiversity (REDD+) benefits. Climate benefits are additional if they would not have been gained without REDD. Such benefits include the carbon stock preserve and the actual emissions avoided from maintaining the forests all of which must be subject to measurement, monitoring, reporting, and verification. In addition, REDD and REDD+ mechanisms must control for leakage, or the relocation of deforesting activities in areas not included into programme. Another challenge relates to the ability of REDD in achieved benefits over time.

Additionality. Within REDD the issue of additionality deals with the fact REDD policy efforts should be targeted to countries, regions or areas where deforestation needs to be stopped. If these efforts are focused on areas where no-deforestation is going to take place then the impact is null (Andam et al., 2008; Pfaff et al., 2008). Overall effectiveness of REDD is compromised since resources are going to waste by financing protection of areas that are not under deforestation threat.

Leakage, in the REDD context implies that efforts to reduce deforestation and forest degradation in any given area would increase deforestation in other region or area. It can be caused for two main reasons: deforestation triggering factors might be relocated to other areas (primary leakage); and/or increases in the market prices of timber, livestock and crops as result of reduced deforestation, which might increase profitability of those activities in areas where they were not economically viable before (secondary or partial equilibrium leakage) (Myers-Madeira, 2008; Wunder, 2008).

To avoid leakage, a country's specific conditions must be well known. This requires identification of the main factors that trigger deforestation so actions to tackle them are properly identified and accomplished. Previous forestry projects have shown that resources can go to waste if implemented without previous knowledge of the causes of deforestation (Chomitz et al., 2007).

Although agriculture and pasture expansion are considered the major drivers of deforestation worldwide, it is necessary to further analyse what main activities (particular crops and pastures) are pushing the forest frontier in and to know the business characteristics of such activities. It would be expected for example that export crops would present a harder burden on forests than auto consumption food production. In this way, the real factors causing deforestation can be directly tackled and an incentivised (Verchot and Petkova, 2010).

REDD effectiveness can be enhanced by decreasing and preventing leakage. This can be done by performing proper monitoring; increasing the scale of REDD application; imposing discounts on carbon credits to reward only net emission reductions, among other actions (Wunder, 2008).

Permanence is another issue surrounding REDD concerns, which are based on the idea that reduced emissions in one year can be reversed in the following year due to forest vulnerability to fires, pests, management and other natural and anthropogenic instabilities. Non-permanence risks can be managed by the use of reserve accounts (a percentage of the credits earned are saved in case deforestation rates increase in a given year); or by defining expiration periods after which carbon credits must be earned again upon new certification (Myers-Madeira, 2008).

Baselines are also thought as determinants of REDD effectiveness. Depending on how those reference levels are to be set, they can potentially affect country participation, the carbon global market, additionality, and resource distribution. For instance, if baselines are set too tight, countries might not participate as the incentives to join are low. If on the contrary baselines are set too generous, the risks of comprising additionality are increased.

The year to determine the baseline would certainly affect countries in different ways. Since the baselines are the criterion to measure avoided deforestation in the future, countries that have a history of high rates of deforestation may turn attractive to implement REDD+ projects that are indeed additional. This is one of the reason the additionally test is still a major subject of debate in the negotiations, in which some countries highlight whether policies in place that avoid deforestation versus building that capacity in countries with high deforestation should be treated differently. There are also concerns that any REDD scheme will overflow the carbon market crowding out other emission strategies.

A less technical aspect that will have strong implications for REDD effectiveness is participation of indigenous and local communities. As it is the case for most REDD features, ensuring participation of indigenous and local communities is case-specific and in many cases will require governance reforms beyond the forest sector. However, participation of these target groups can be enhanced following some main principles (Angelsen et al., 2009):

I. Defining land, resources and ecosystem services rights.

II. Safeguarding participation in REDD decision making.

III. Making REDD part of a long-term development policy.

IV. Ensuring direct access to financial resources.

Some argue that point 2 of the list urges proper participation of local and indigenous communities into the Conference of the Parts (COP) and the Kyoto protocol. However, this can further limit the ability of these meetings and mechanisms to deliver concrete and verifiable agreements. As alternative proposals are mentioned the creation of an appeal system that allows submitting complains when a party has not abided to the agreement; and granting rights and protection mechanisms at the international level to non-national groups (Angelsen et al., 2009).

In addition to those main principles, participation of local and indigenous communities can be improved with the use of existing international treaties that preserve the rights of such populations.

2.1.2 Cost-effectiveness or other means of economic efficiency

Cost-effectiveness implies that emission reductions must be achieved at the lowest possible cost (Angelsen, 2008). Pagiola and Bosquet (2010) assert that global cost estimations of REDD are not enough for countries to develop their specific policies to reduce carbon emissions. For countries to benefit from REDD, they must find their own most cost-effective alternatives per unit of carbon reduced[3]. Those costs

[3] *This is the remit of POLICYMIX except that “effectiveness” of main concern is biodiversity conservation.*

will mostly depend on each country's own characteristics such as its agro-ecological, economic, and social conditions.

A country wise analysis of the specific costs of REDD would also allow countries to estimate their own supply function and therefore, be able to estimate the reduction amount they are able to provide at any given price per CO_2 ton reduced (Pagiola and Bosquet, 2010). An important feature tested through empirical estimation of supply curves is the existence of increasing marginal costs, which indicates that reducing emissions gets increasingly more expensive (Lubowski, 2008).

However, for the cost estimation to be useful, total cost associated to emission reduction must be considered. This requires fully taking into account three cost categories: opportunity cost, implementation cost and transaction costs (Pagiola and Bosquet, 2010; Lubowski, 2008); although the three are not necessarily additive. Furthermore, additionality and leakage must be taken into account since they are likely to reduce efficiency and/or increment costs. Kindermann et al. (2008) for instance assert that leakage estimates in forestry projects vary between 10 and 90%.

Opportunity cost quantifies forgone benefits as deforestation or forest degradation is avoided (pasture and agricultural land or timber product extraction for example). It is considered by some the cost item with highest share into total REDD costs (Pagiola and Bosquet, 2010) and as such, is the focal problem when estimating REDD cost.

A good estimation of opportunity costs will shed light on the reasons that explain deforestation and forest degradation at the national and local levels. It will also identify who would loss and who would win when the mechanism has been launched, which brings up the issue of equity and social impact discussed in the following section (Pagiola and Bosquet, 2010).

Inappropriate estimations can give the wrong market signals that can potentially hinder the success of REDD. For example, it has been estimated that total costs of REDD are highly sensitive to the assumptions made about revenues of alternative land uses, especially related to agriculture production. Total cost of REDD in eight countries[4] increased US$ 1.5 billion with respect to the initial estimation made due to increases in the price of palm-oil (Grieg-Gran, 2008).

Implementation costs are all the expenses related with actually avoiding deforestation and forest degradation. These includes specific actions aiming at protecting forest such as plans to alleviate pressure on forest, costs associated to preventing forest uses other than conservation, and, the institutional capacity building to implement REDD at the country level (Pagiola and Bosquet, 2010).

Transaction cost*:* this category of costs is not related to the actions taken to reduce forest degradation and deforestation; instead it accounts for the activities needed for the programme to be transparent and credible such as monitoring of emissions, verification of the CO_2 emissions actually reduced, transactions with the institution in charge of payments, etc.

Another important factor to consider when estimating REDD cost is for whom the costs are being estimated. If estimations are for the country as a whole, then resources must be priced at their social value while, if costs are estimated for individuals or individual groups, resources must be valued at market prices (including distortions such as taxes and subsidies) because they represent the real opportunity costs faced by individuals (Pagiola and Bosquet, 2010).

[4] *The analysis included Brazil, Indonesia, Papua New Guinea, Cameroon, Congo, China and Costa Rica.*

Regarding programme scale, Angelsen et al. (2008) state that national approaches tend to be more efficent due to economies of scale realised in monitoring, reporting and verification activities. In addition, national programmes are likely more capable of controling for domestic leackage. As stated by Wunder (2008), "*leakage can occur whenever the spatial scale of intervention is inferior to the fill scale of the targeted problem.*"

Initial assessments of the potential costs of REDD seem very optimistic. REDD has been regarded as a cheap way to reduce carbon emissions based on the fact that around 20% of total carbon emissions come from deforestation and forest degradation (DD) and therefore, reducing DD emissions would be relatively easy. Additionally, it has been argued that most DD is only marginally profitable having low opportunity costs and consequently, it would be relatively inexpensive to compensate individuals to avoid DD (Angelsen, 2008).

Other analyses have concluded that initial evaluations of pilot REDD projects were based on simplistic assumptions and that REDD implies a complex institutional development (Lubowski, 2008; Blackman, 2010; Pagiola and Bosquet, 2010)[5]. As a result, it might not be as cost-effective as originally thought. For example, earlier evaluations of REDD assumed 100% additionality, no leakage and negligible transaction costs (Blackman, 2010). As Pagiola and Bosquet (2010) assert, appropriate cost estimation of REDD requires a good analytical understanding of all variables affecting REDD, not only at the individual but also at the national level. Furthermore, costs associated to REDD are not only related to its implementation but also to the institutional development required at the national and international levels and to private costs.

2.2 Economic and social impacts

When talking about social impacts, equity is one of the most important issues to consider. Because of the multistage nature of REDD, its equity implications must be analysed at two levels: international and country-wide. Equity at the international level requires defining a plan that allows low income countries to participate on the programme, even if they do not have the institutional structure to implement a national programme (Angelsen et al., 2008).

At the national level, opportunities to participate in the programme must be ensured to all possible applicants, independently of their social and economic status. In this sense, subnational and nested approaches are considered more flexible and able to respond to specific contexts faced by minority groups (Angelsen et al., 2008).

There must also be an equitable distribution of benefits from REDD among stakeholders, communities and individuals. Fairness can be ensured through several mechanisms such as transfer of forest tenure (to indigenous people with no formal rights to land) and distribution of carbon rights and benefit-sharing agreements (Schwartzmann et al.; 2008; Sikor et al., 2010).

Specific analyses in different countries are not categorical respect to the impact of payments for environmental services (PES) on poverty. One of the main issues in this regard are the permanency variable. In other words, although incomes are proven to be complemented by PES programmes, it is not possible to conclude whether poverty is been alleviated since most PES programmes have a short-term life spam or, financial resources are not guaranteed to stay forever.

[5] *For the purpose of POLICYMIX, the study will focus in the relevant complexities of Costa Rica and Brazil.*

The impact of REDD programmes will also depend on the way they are designed, implemented and on the activities they involve. If the programme includes participation of local individuals, communities and stakeholders, the likelihood of it having a social positive impact is high. In addition, equity decisions do affect the social impact of a given project. Usually a decision must be taken regarding whether to finance relatively few individuals with high payments or to include more participants at the expense of the payment amount.

An example of PES evaluations in the REDD framework is the one performed in Mozambique. It was determined that the project was able to supplement family's incomes in the short run (between 2004 and 2008) but it was not enough to eradicate poverty in most beneficiaries (Jindal, 2010).

There are concerns that REDD might harm indigenous groups (i.e. Boucher, 2008). However, the argument is highly debatable as long as land tenure and credit rights are enforced. Indeed, an analysis of 13 case studies in Latin America, South Asia and Africa did not find evidence that PES programmes have had a negative effect on livelihoods and equity (Bond et al., 2009) but also acknowledged that at the larger scale likely with REDD, the PES experience might not be relevant.

The implementation of protected areas as REDD efforts can also have important economic and social effects. Sims (2010) finds that protected areas are associated with lower levels of poverty and higher levels of consumption in Thailand; Andam et al. (2010) also finds that protected areas are associated to lower poverty levels in Costa Rica. Furthermore, Robalino and Villalobos-Fiatt (2010) argue that protected areas in Costa Rica lead in certain areas to higher wages and lower unemployment rates[6].

2.3 Institutional context and requirements

Institutional and political variables can constraint REDD effectiveness and economic efficiency the same way they are responsible factors of deforestation in some countries. As it has been mentioned before, a country's capacity to embrace into a national programme is likely to be limited by its management and monitoring capacities (Boucher, 2008). Investments aiming to improve governance structures are required to increase REDD effectiveness. Strengthening governance requires defining rules, rights and institutions at all levels: national, local and civil society (Bond et al., 2009).

A sophisticated and well-defined institutional framework can help achieve the REDD objectives. A comprehensive institutional structure for REDD governance at the national level is in part determined by international law and the relevant Multilateral Environmental Agreements (MEAs)[7]. Moreover, there are unilateral decisions, bilateral agreements, and regimes that may affect institutions in other countries interested in implementing REDD programmes[8].

Institutional development needs are country specific, responding to each nation's own laws and institutions. This is evident when the national legal framework of the Democratic Republic of Congo

[6] *For those interested in this issue, refer to the review on 'direct regulation' (see Schröter-Schlaack and Blumentrath, this report).*
[7] *For example the UNFCCC and its protocol; the Vienna Convention for the Protection of the Ozone Layer and its protocols; certain agreements under the World Trade Organisation; the Convention on Biological Diversity; the United Nations Forum on Forest, etc.*
[8] *Fox example in the USA the American Clean Energy and Security Act of 2009 and the Clean Energy Jobs and American Power Act of 2009 (see Sheikh and Gorte, 2009). In the European Union the EU Action Plan for Forest Law Enforcement, Governance and Trade (FLEGT).*

(DRC), Indonesia and Brazil are compared. While in Brazil there is base forestry legislation that could serve as a REDD entry point, such regulation is inexistent in DRC. Brazil and Indonesia have experience at some degree with forest certification, putting them a step ahead of DRC; a legal agenda is emerging in Brazil to safeguard indigenous group's interests, which is inexistent in the other two countries. In addition, there are also differences in the levels of law implementation among the three countries. While Brazil is improving in this respect, Indonesia's capacity is weak while in DRC there is not even an operational framework to be enforced (Bond et al., 2009).

Three main institutional aspects must be enhanced at the national and subnational levels: clarification of forest-dependent communities' rights; facilitation of equitable sharing of benefits and promotion of sustainable forest management. To achieve those goals, the following actions are needed (Bond et al., 2009):

- Dialogue between all stakeholders to design national and local policies and institutions needed for REDD implementation.
- Integration of REDD into national and local policies and institutions.
- Reform of national forest laws.

A successful institutional structure for REDD should be adaptive to developments that arise from new scientific findings and policy outcomes. National governments should clearly define and guarantee land ownership and use rights, the share of benefits, processes to implement MRV, access to information, transparency, and public participation. There are at least three crucial factors for success in the implementation of a REDD instrument; (i) compatibility between REDD and national legislation to foster effectiveness; (ii) law enforcement, to reduce political and financial investment risk; and, (iii) and accountability, to better deal with competing interests among stakeholders (Costenbader, 2009).

In this sense, chances are that there would be institutions that can hamper effectiveness of a well-defined REDD strategy. In the most common scenario, national development plans may incentivise legal economic activities that promote deforestation and forest degradation such as agriculture subsidies and forest use concession rights. Under the principle of law enforcement explained above, the legality of carrying out such activities will hamper the effectiveness of REDD activities.

Here, the capacity of adapting institutional mechanisms is key to maximise programme's success. On the other hand, policymakers must be sensitive to certain traditional practices that reduce the carbon stock such as traditional farming, slash-and-burn of forest, and charcoal production in ways that do not constrain the public acceptance of the REDD programmes.

A related issue is the definition of the level of REDD application. Whether actions are to be national or project-based will affect hot issues as additionality and permanence. For example a national based policy would be able to account for leakage, liability, permanence, and the scale would be large enough for achieving significant reductions of emissions. Nonetheless, such an approach would have to deal with bureaucratic procedures and lack of institutional capacity in some cases. Presumably, project-based REDD policies are easier to implement but their potential impact on deforestation and emissions reductions is smaller. The idea of implementing hybrid policies is an interesting one that would have to address these elements in its design (Myers-Madeira, 2008).

3 Some experiences from pilot REDD project

Lessons from forest-based mitigation projects show that design and implementation capacity are key features in REDD projects. Some of the major problems revolve around weak social objectives, communication deficiencies, time constraints, and limited local benefits (Boyd et al., 2007). Some of the fundamental lessons learned from these projects are the crucial role effective project-administration and risk management plays. In this since, it is important that project developers understand the local context in terms of history and politics. The linkage between social, environmental, and economic objectives has to be explicitly defined at all stages of the project. In addition, guiding regulations, organisational capacity, and appropriate decision-making processes have to be skilfully conceived in the policy design.

This section presents a few case studies implemented as pilot REDD projects that have tried to address the most important elements regarding the legal and technical challenges for REDD project design[9].

3.1 Baselines, leakage and permanence

In 2004 the Ankeniheny-Zahamena-Mantadia Biodiversity Conservation Corridor (CAZ) and Restoration Project ("Mantadia project") was created by a partnership between the Government of Madagascar and a network of national and international non-profit organisations to decrease forest loss in Madagascar.

The corridor creation's goal was to reduce deforestation on approximately 420,000 hectares under the REDD flag, which is complemented with an Afforestation/Reforestation (AR) component. The AR component, known as Tetik' Asa Mampody Savoka (TAMS) will eventually restore forest cover on approximately 3,000 hectares of degraded lands. It is expected that approximately one million tCO_2e will be sequestered over the 30-year project lifespan.

Both permanence and leakage issues are addressed in the project design by including legal protected area status and community development activities. In the short term the project is expected to undergo at least three validation and verification processes: (i) under the Voluntary Carbon Standard (VCS) for the REDD component; (ii) VCS and Clean Development Mechanism for different portions of the AR component; and, (iii) Climate, Community and Biodiversity Standard for both components.

The project developers are using the 2008 BioCarbon Fund of the World Bank (BioCF) methodology to calculate the baselines and expected project carbon. This tool combines two basic components to predict future emissions from deforestation in the business-as-usual (baseline) case: The first consists in quantifying the projected levels of deforestation based on observed historical rates. The second comprises a spatial land-use change model to predict where deforestation will occur based on the relationship between past deforestation and certain drivers of deforestation (e.g. distance to roads, terrain slope, and distance to markets). The model of expected future deforestation was created using data on deforestation and its driver, and explicit forest cover from 1990, 2000, and 2005. The strength of the model to predict future deforestation was tested by forecasting the forest cover in 2005 and comparing the results with real forest cover data from satellite image for that year. Then the model was run forward out to 2035 to predict the location of future forest cover changes inside the project area. Up to date there are not reported impacts of the project.

[9] *For a detailed account of each project see The Nature Conservancy et al. (2010); The Nature Conservancy (2009); Boyd et al. (2007); Börner and Wunder (2008).*

3.2 Permanence, measuring, and monitoring

The Makira Forest Protected Area (Makira project) was established in 2001 by the Madagascar Ministry of Environment, Forest and Tourism, in collaboration with the Wildlife Conservation Society to protect 372,470 hectares of land; the largest remaining contiguous tract of low and mid-altitude rainforest in eastern Madagascar. The goal was to reduce human threats to the forests while engaging local communities in the management of the protected area. Deforestation by slash-and-burn for agriculture, hunting, and exploitation of timber and non-timber forest products were some of the major drivers of deforestation in the region.

The project design addresses the issues of permanence and leakage through the legal definition of a protected area, sustainable land management for community development, and granting legal property rights to locals. Monitoring of adjacent areas is done by using satellite image and surveys. The project is currently undergoing validation under the Voluntary Carbon Standard (VCS) and Climate, Community and Biodiversity Standard and also plans to verify carbon benefits through VCS.

3.3 Leakage, Standards, and Verification

The Noel Kempff Mercado Climate Action Project (NK-CAP) was one of the world's first large-scale REDD projects addressing deforestation from conversion to agriculture by local communities and degradation from logging activities in timber concessions. It comprises US$ 8.25 million in carbon financing and possible additional financing with sales of carbon offsets by the Government of Bolivia. In 2005 it became the first REDD project to be verified by a third party using the Kyoto Protocol's CDM standards.

This Project is an example of how well-designed REDD projects can result in real, scientifically measurable, and verifiable emissions reductions with important benefits for biodiversity and local communities. By its implementation, over a million metric tons of verified CO_2 emissions were avoided between 1997 and 2005. Furthermore, it is estimated that a total of 5.8 million metric tons of CO_2 emissions will be avoided over the 30 year lifespan of the project.

Noel Kempff Mercado National Park is part of a UNESCO World Heritage Site which preserves rich biodiversity and provides sustainable economic opportunities for the local population through community forestry and ecotourism. This project helped indigenous communities achieve legal status by obtaining official land title (The Nature Conservancy, 2009).

To counteract the potential of leakage the financing schemes in REDD policy design may need to have a broader spatial coverage of all areas potentially at risk which will make the implementation of REDD activities more costly than widely believed (Börner and Wunder, 2008; Blackman, 2010).

3.4 Scale and Scope

The Berau Forest Carbon Programme (Berau Programme) is located in northeastern Borneo. The area is heavily forested and rich in wildlife but is threatened by the expansion of commercial logging and oil palm activities. The local government, the Government of Indonesia in partnership with The Nature Conservancy are developing a forest carbon programme that addresses deforestation and forest degradation in an area that covers over 2.2 million hectares. The programme includes working with logging concessionaires to implement Improved Forest Management (IFM) practices for wood production while reducing forest damage and high levels of carbon emissions. The programme will also create a model to shift oil palm production to areas with degraded lands. The issues of illegal logging and clearing for agriculture is addressed by working with local communities to manage the new existing protected areas so they do not lose carbon stocks. It is estimated that the programme will avoid the emission of 10 million metric tons of CO_2 over five years.

A lot of the data and results for these case studies are still being tested. Since REDD projects must be evaluated at least every 10 years, it would be interesting to see the outcomes of these projects in the near future. What is important to highlight from this section is current technology and methodologies can provide credible information. This in turn helps to reduce uncertainty regarding the legal and technical challenges for REDD projects and policy design.

4 The role of the instrument in a policy mix

REDD can be combined with several other economic instruments for forest conservation and management. Ring et al. (2010) mention the potentialities of REDD complementing, or being complemented by two innovative instruments: fiscal transfers and tradable permits. The authors argue that those instruments are highly cost effective and can potentially tackle the negative externalities of land development and internalise the positive externalities of conservation measures and protected areas.

The combination of REDD with fiscal transfers is hypothesised to be straightforward in the sense that there are already existing carbon markets functioning at the several levels. The drawback of the possible mix of both policies strikes in the possible trade-offs existing between carbon stocks and biodiversity conservation (Ring et al., 2010). The policy mix must therefore be able to find the optimal combination of both objectives or at least, define the minimum acceptable level of any one of them while the other is maximised.

Although fiscal transfers have traditionally been used as a mean of mobilising resources from national governments to lower-scale governance institutions, they have been proposed as a way of relocating international funds to national and sub-national instances to finance efforts for biodiversity conservation and forest conservation and management. A desirable aspect of fiscal transfers is that it can finance the development of the required institutional level required for REDD to be implemented. However, excessive cash flows can hinder conservation efforts as they might incentivise and/ or exacerbate corruption (Ring et al., 2010).

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Forest Certification:
A Voluntary Instrument for Environmental Governance

Karin Kaechele, Peter May, Eeva Primmer and Grit Ludwig

Summary

Forest certification acts as a bridge between market regulation and environmental governance by furnishing specific criteria in response to consumers and buyers' demands that production practices ensure forest integrity and resilience. In this sense, voluntary certification acts as a non-state arbiter of conformity with quality and performance criteria in achieving socio-environmental goals. This chapter seeks to analyse the limitations and perspectives that forest certification may contribute toward biodiversity conservation as part of a broader policy context. The study describes a range of forest certification initiatives around the world, how they operate and who are the primary stakeholders involved. It also discusses key elements such as the institutional context and requirements for the instrument to be effective for biodiversity conservation and to minimise the negative socioeconomic and environmental impacts of forest resource utilisation. It concludes with a brief analysis of the role of forest certification in a mix of command-and-control and economic instruments.

1 Certification of forestry practice

1.1 Conformity of production practices with certification criteria

Certification is a procedure by which a third party gives written assurance that a product, process or service conforms with certain standards. It is also a guarantee of origin that is used to orient the consumer in product choice, with some form of added value, usually derived from environmental integrity and/or social fairness. Often certification schemes are considered non-state and market driven but typically they involve public sector actors and assume conformance with state regulation. The degree to which certification reflects autonomy and external verification varies. According to Conroy (2007), certification processes may be classified into three distinct stages, characterising different degrees of autonomy in verification and monitoring:

i) Codes of conduct or declarations of good intentions adopted unilaterally on the part of companies, known as "*First party certification"*. These have the advantage to call attention to the consumer regarding the form of production (socio-environmental footprint of the purchase), and not only what is being produced (price, quality). However, these instruments carry the risk of being used inappropriately as certificates or testaments of sustainability simply to avert criticism. Due to their unilateral character, these instruments offer limited credibility on their own and are mere expressions of the adopting organisations' willingness to enter a market and compete on the grounds of social acceptability and environmental integrity.

ii) Initiatives of business groups and associations in an activity adopting defined labels and certificates. This type of initiative has been called "Second party certification" since it includes the intervention of another actor, an association, besides the company itself. The certification schemes advocated by some states and groups of states can be considered to fall under this category, as the state is somewhat external to the certified producing companies (Cashore et al., 2005). It similarly allows the consumer to focus attention on aspects of production or on how internal corporate operations are conducted. Because they are verified by an actor external to the company, these certifications have a bit more credibility, but this depends on the credibility of the independence of the external actor with respect to the specific company under review.

iii) "Multi-stakeholder" initiatives, also called "Third party certification" are inspired by the "stakeholders" theory enunciated by Freeman (1984)[1]. An important characteristic in this approach is that the "third party" is a non-state private regulator (Cashore et al., 2005). This approach seeks to manage social and environmental responsibility in response to concerns expressed by numerous interested or associated parties. It assumes that the company needs to invest in engagement with its "stakeholders" not only for ethical reasons but also to access and maintain a position in the market, achieve and maintain reputation and improve competitiveness. From this perspective, a company worthy of a certificate is one that is attentive to its stakeholders' concerns. Thus, this third category of certification instruments presents a completely different way of dealing with environmental responsibility, in which dialogue and interactions with and among stakeholders are of paramount importance. Certification thus challenges the traditional state-centred idea of regulation, as it shifts control and power from the public sector to the auditing third party and to final consumers. In doing so, it establishes a basis for consumer confidence, expressed in legitimation of third-party labelling.

1.2 Initiatives toward forest certification around the world

The movement in favour of forest certification began at the end of the 1980s, with commercial boycotts by consumers in Northern countries against logging of tropical timbers originating from deforestation. In this context, European and North American tropical wood consumers, concerned with their long-term business prospects formed an alliance for protection of tropical forests – the *Woodworker's Alliance for Rainforest Protection* (WARP), and published a "Good Wood List" in an effort to protect wood suppliers derived from "good management". In 1993, representatives of NGOs, suppliers and buyers of wood met in Toronto, initiating a process that led to the creation of the "Forest Stewardship Council" (FSC). In response to the lack of criteria to define what constituted good forest management practice, three international chambers, representing commercial, social and environmental concerns instituted 10 principles and a rigorous body of subsidiary norms (Azevedo-Ramos et al., 2006).

The FSC gained popularity also among northern timber producing and processing actors and states, partly spurred by the general trend of searching for forms of governance alternative to state control in both North America and Europe (Cashore et al., 2003; Rametsteiner and Simula, 2003). In some cases, the evolution of the FSC stimulated fierce dialogue between different forestry regimes and sustainability standard controlling systems, with forest industries and timber producers dominantly favouring national certification schemes at the outset (Cashore et al., 2003). With further legitimacy pressure from markets and environmental NGOs, many northern timber processing and consuming countries dependent on international markets eventually also adopted the FSC system (Cashore et al., 2003; Gulbrandsen, 2004). In countries such as the USA, Finland and Norway, where landownership was predominantly small-scale

[1] Stakeholders are those groups that affect and/or are affected by the organisation and its activities. These can include, but are not limited to: landowners, administrators, functionaries and labour unions, clients, associates, business partners, suppliers, competitors, government and regulatory agencies, the electorate, non-governmental organisations/non-profits, pressure groups and opinion leaders, and local and international communities. In fact, the definition of the stakeholders' boundaries becomes a determining factor (Bodet and Lamarche, 2007). As J. Samuelson recalls: «...and if you discern who your stakeholders are, it is very likely that they will do this for you... » (Samuelson, 2008).

and the forest sector was relatively powerful, the national certification systems maintained their position of dominance over the FSC.

The European national forest certification schemes are grouped under the Pan-European Forest Certification (PEFC) scheme, which functions as a rather open umbrella, but which is internationally powerful due to its large geographical coverage. Clearly, throughout the disputes, the FSC has remained popular among environmental NGOs, and has continued to attract companies and regimes that are most sensitive to market and social legitimacy pressures.

Internationally, voluntary forest certification has evolved since its inception in the 1980s, and now embraces a range of systems in operation which are in competition. Among these, the principal labels include:

- **Forest Stewardship Council** - FSC, is an international non-governmental organisation, founded in 1993, which accredits certifiers throughout the world, guaranteeing the certified parties obey strict quality norms. Certifiers undertake a methodology based on the FSC Principles and Criteria (P&C; see listing in Annex), adapting themselves to the reality of each region or production system. The FSC has decentralised into a number of national or regional initiatives, which have developed their own respective P&C, adapted to local technical conditions, forest resources and legal context.
- **Programme for the Endorsement of Forest Certification Schemes - PEFC** (originally Pan European Forest Certification). The PEFC Council was created in June 1999, also of voluntary nature, based on its own criteria defined in the Helsinki and Lisbon Conferences of 1993 and 1998, respectively, on European Forest Protection. A primordial objective of this system is the recognition of different systems operating in the European Community. However, PEFC schemes embrace those adopted in other regions as well. For example, the Brazilian Cerflor system (see below) has received provisional recognition by PEFC.
- **A range of diverse national systems** (Sweden, Finland, Norway, Germany, the UK, United States, Canada, South Africa, Indonesia, Malaysia, New Zealand, Chile, Austria, Ghana, Belgium and others).
- In Brazil, the **Cerflor** system – the Brazilian Programme for Forest Certification, was conceived by the Brazilian Silvicultural Society (SBS) in 1996, though it only began to operate nearly a decade later. Cerflor was created in partnership with sectoral associations, research and training institutions, NGOs and with the support of several government agencies, including the national standards institute. It differs from the Brazilian FSC standard in some respects, having somewhat more relaxed criteria for social concerns. However Cerflor enjoys considerable credibility in part due to its PEFC recognition.

According to Purbawiyatna and Simula (2008) ‘almost two-thirds (65%) of the world’s certified forests (in 22 countries) carry a PEFC certificate, while the FSC’s share is 28% (in 78 countries); the remaining forests are certified solely under national systems. Most of the certified forests in the tropics are FSC-certified.’ The FSC had more than triple the number of products under chain-of-custody certification in 2007 as compared to PEFC certifiers.

Approximately 8% of global forest area has been certified under a variety of schemes (FAO, 2010) cited in ETFRN (2010). One recent estimate suggests that approximately one quarter of global industrial roundwood now comes from certified forests. Most of these advances have occurred outside the tropics: less than 2% of the forest area in African, Asian and tropical American forests are certified. Most certified forests (82%) are large and managed by the private sector (ETFRN, 2010).

1.3 The forest certification process

Certification is a voluntary process through which a forestry enterprise is evaluated by an independent organisation – the certifier – permitting that the firm's compliance with environmental, economic and social concerns is verified in accordance with the P&C of the particular certification system being applied.

The process can be broken down into its principal stages:

- **Initial contact** – the forestry operation enters into contact with the certifier.
- **Evaluation** – A general analysis of management, documentation and field appraisal. Its objective is to prepare the operation to receive certification. In this phase public consultations may be arranged, so as to obtain feedback from stakeholders.
- **Adaptation** – After evaluation, the forestry operation should adapt non-conforming practices if these appear.
- **Certification of operation** – the forestry operation receives the certification. In this stage the certifier prepares and makes available a public summary.
- **Annual monitoring** – After certification the operation is monitored at least once each year to maintain the certification.

1.4 Actors involved in forest certification processes

Usually the actors involved in certification processes represent timber producers, civil society organisations, researchers, industries and, to varying degrees and with varying roles, also government authorities. When there is a community forest management plan, the local government is involved. The graph below shows how the actors are distributed within two distinct types of forest certification applied in Brazil: FSC and Cerflor. In general, the objective of the group is to develop P&C that fit each region for native forest management or forest plantations. Public consultation and the involvement of local communities are also part of the process of certification.

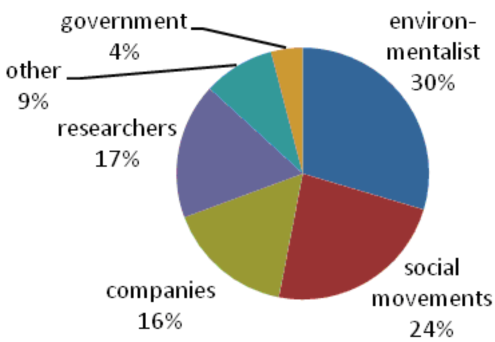

Figure 1. Actors involved in FSC certification

Source: adapted from Greenpeace (2002)

As is clear from the graph presented (Figure 1) the FSC process seeks to attain a balanced representation among social actors, with the exception of government, while Cerflor is heavily weighted toward corporate actors, researchers and government, with little or no representation of environmental or social civil society organisations (Figure 2). This pattern has been found to be typical in comparative research across many regimes where FSC competes with local schemes (Cashore et al., 2005).

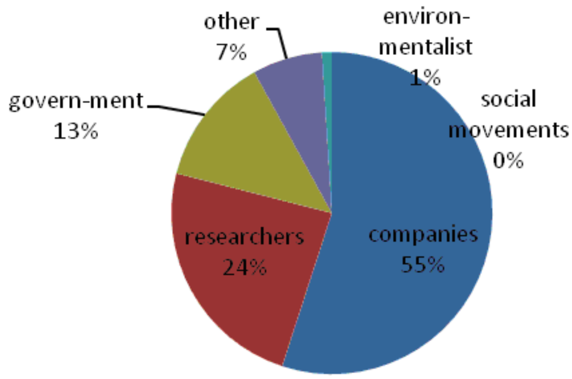

Figure 2. Actors involved in CERFLOR certification

Source: adapted from Greenpeace (2002)

1.5 Baseline

The certification process is generally progressive. The forest enterprises at first meet basic requirements and over time improve performance based on the auditor's recommendations.

Certification is also an adaptive and gradual process that permits certain flexibility of rules and criteria. As an example, FSC cites its adoption of the SLIMF (Small and Low Intensity Managed Forests) procedures for progressive adaptation to general P&C. The simplified SLIMF audit procedures can be used in enterprises, such as communities, small farmers and businesses that manage small areas or low intensity forests. It can also be applied to non-timber forest enterprises, provided they are not based on plantations. Using the SLIMF method, the audit process simplification reduces costs and time needed for evaluation. In this type of technical evaluation, although the same FSC standards and rules are applied, the simplification in the process enables small producers to participate in certification appropriate to their scale and special needs.

1.6 Monitoring, reporting and verification (MRV)

Verification is a key tool of socio-environmental responsibility initiatives; verification involves checking compliance of procedures with criteria. It covers inspections and tests performed at different points of the production chain or on the whole process.

The strictness of the verification phase ensures credibility of the initiative; the agency in charge of this should be independent with no financial and corporate ties to the initiative. Thus, the agency's independence ensures autonomy and impartiality of the verification process.

The transparency of verification is crucial. The information available should include: the methodology used, the points checked as well as the positive and negative results of implementation procedures and criteria.

Finally, a socially responsible verification should feature a conflict resolution mechanism that makes it possible for the players, regardless of whether or not they participate in the initiative, to denounce actions not complying with their commitment.

2 Performance of certification in environmental governance

2.1 Effectiveness for biodiversity conservation and the provision of ecosystem services

Forest certification should assure that the timber used in a given product originates from forests managed and processed in accordance with sustainability principles in a fashion that is simultaneously ecologically sound, socially just and economically viable. Additionally, conformance with standing legal codes is a universal certification requirement. In this way, certification entails both public and private regulation characteristics (Cashore et al., 2005; Potoski and Prakash, 2004).

Effectiveness in forest certification refers to the amount of native forest that is managed and not clearcut or the proportion of forest managed according to sustainability criteria, but also refers to how local actors are engaged in the forest management enterprise.

Some authors (Brotto et al., 2010; Gullison, 2003; IMAFLORA, 2009; Price, 2003) agree that certification has helped to improve management practices and to conserve forest biodiversity within certified forests in the tropics as well as in other regions where state governance of forest management has faced challenges (Cashore et al., 2005; Keskitalo et al., 2009; Rametsteiner and Simula, 2003). However, the true extent of conservation benefits remains unknown due to a lack of rigorous and independent information. Many agree that certification is not equivalent to full conservation and point at the limitations of certification in reducing deforestation rates. Some examples of biodiversity assessment on forest certification are described below:

In 2009, IMAFLORA, the Institute for Forest and Agricultural Management and Certification, a SmartWood certifier based in Brazil conducted a study of the impact of FSC in a planted forest in southern Brazil and an extractive community in Acre state, which revealed that FSC Forest certification resulted in positive impacts regarding the environmental aspects assessed, such as natural resource conservation, forest management, and its contribution toward conservation of flora and fauna and the water resources of natural ecological systems.

In its evaluation in Acre extractive communities where forest management is performed, positive impacts were found to have resulted from FSC certification actions. The survey found that the use of fire for clearing planted areas is a common practice in all Extractive Settlements areas studied. However, the findings indicated that slash and burn clearing is less harmful in the certified communities than in the non-certified ones due to the forest care requirements of the certification method. Hunting is widespread both in certified and non-certified areas. However, with respect to the care taken during this practice, the survey found a significant difference between the certified and non-certified groups: of the certified areas, 87% reported the use of measures for protecting wild animals, compared with only 44% of the non-certified groups. The measures cited by the certified communities were hunting only when food is needed and using no dogs in hunting. Beyond these measures, they also reported the use of some others, such as hunting season calendars, not killing animals nursing their young and preserving trees that provide food for such animals (IMAFLORA, 2008).

When analysing planted forests in southern Brazil, natural resource conservation was assessed by the following actions of the enterprises sampled: environmental licensing, legal reserve[2] registration, control of invasive species in Permanent Protection Areas (APP), reforestation with native species and studies of fauna and flora. Furthermore, IMAFLORA investigated signs of forest conversion in the enterprise (replacement of forest fragments for agricultural, livestock, forestry, etc.) and the proportion of native forest remaining on the property. Impacts of FSC certification on natural resource conservation in the enterprise studies were evident. The certified enterprises control weeds in APP, initiate and maintain fauna and flora studies and do not carry out any forest conversion aside from that necessary to observe effects in a control site.

Due to the constant changes in legislation, the certified enterprises presented mechanisms for monitoring the environmental legislation and securing or being in the process of obtaining environmental licenses and legal reserve registrations. IMAFLORA also examined evidence of riparian forest uses in APPs, as well as care in the forest management in the surrounding areas. According to the enterprise representative's testimony, there was certification impact on the different treatment given to the management of the areas close to APPs: sensitive natural area delineation, pre and post-harvest evaluation in the buffer areas, targeting the harvest and identification of trees for bird conservation.

The impact generated by the certification actions in extractive communities is low (IMAFLORA, 2008, 2009; TEEB, 2010). However, there is little quantitative evidence regarding the long-term impacts of certification on biodiversity and the environment. FSC certification positively impacts forest planning and inventorying, silviculture, biodiversity protection, and monitoring and compliance.

In a recent publication organised by ETFRN (2010), biodiversity benefits from forest certification are explored. Below some examples are cited from this recent survey.

Price (2010) describes research conducted in Bolivia and in the Brazilian Atlantic forest evaluating the impact of certification on those forests. Price concluded that the rate of forest loss in the FSC-certified forests was lower than that observed in some of the country's national protected areas. He pointed out that one reason for this is that FSC standards require compliance with legislation. This compliance, along with the remaining rigorous requirements of the FSC standards are more effective in conserving these native ecosystem remnants than the delimitation of protected area status, since often such delimitation only creates 'paper parks'.

Brotto et al. (2010) assessed certification's effectiveness on biodiversity conservation in the Peruvian Amazon, and the economic results arising from such certification. A relationship is drawn between certification and pressures for conversion of native forest into pasture or agriculture land. Since such pressures are high, the opportunity cost to maintain a forest is also high, while the cost for certification is double that of the opportunity cost of retiring land from agriculture. The authors conclude that where a REDD+ project is associated with forest certification, the impact on biodiversity conservation was higher because landowners received a premium on their products.

Gullison (2003), in assessing the overall international experience with FSC considers that certification can contribute to biodiversity conservation, but that the incentives offered by certification are insufficient to

[2] *Legal reserve is the share of native vegetation rural properties in Brazil area required to preserve as part of the "social function" of property, in accordance with the national Forest Code. In the Mata Atlantica rainforest and in the savanna this share is 20%. In the Amazon biome, if the property is located in the forest this percentage is 80% and in the savanna 35%. The Forest Code is currently the object of efforts to undermine its protection of remaining private forestlands by rural landowners.*

prevent deforestation, and the volume of certified forest products currently on the market is too small to significantly reduce logging pressure on High Conservation Value Forest (HCVF). He adds that FSC made great contributions to protection of native forests in temperate countries but in tropical forests very little progress has been made. He concludes stating that industrial logging can produce direct benefits such as avoiding deforestation or improving the value of managed forests, and also indirect effects such as providing alternative timber supplies to those from HCVF.

2.2 Economic and social impacts

In general, the most beneficial impacts found with regard to forest certification were economic and social in character, while the most negative refer to certification cost. Although global demand is growing for certified tropical timbers and other forest products, the intensity of investment, continued difficulties in licensing and transport, unclear land tenure as well as conflict with competing land uses at the frontier, imply that the overall effect of certification has not been to dramatically enhance sustainability at a sectoral level, especially in the Brazilian Amazon. Nevertheless, embarking on a certification strategy in most cases can consolidate the bargaining position of certified timber enterprises with their buyers, as well as providing potential economic advantages (May, 2006).

In general, forest management activities are costly in terms of financial and operational aspects and require those involved in the extraction sites to have high technical capacity in terms of forest inventory, cutting techniques, harvesting and skidding. For this reason, community forest management often must rely on external agencies and the effectiveness of forest management is limited. The cost of FSC certification is seen as exorbitant (US$ 50,000 – 150,000 depending on enterprise scale), which is especially problematic in developing economies (Schepers, 2010). In general, certification can place insurmountable requirements and costs on communities and small-scale actors, and therefore increases the relative power of large scale operators (Klooster, 2005). With regard to the direct costs that result from forest management certification, there is evidence that certification in the tropics is more costly than in temperate or boreal forests for two reasons: First, non-tropical forests are less complex and thus require lower auditing time and preparation, and second, temperate and boreal forests often already have some well-established management procedures in place. Consequently, raising management standards to the required level is less costly. Investors from industrialised countries are usually accustomed to a dense and strict regulatory environment and hence it may be easier for them to comply with rigorous certification criteria (Pattberg, 2005). Considerable cost differences for certification between developed and developing countries have been identified by Gullison (2003): certification costs for large forestry companies in the United States or Poland stand at US$ 0.02 to 0.03 per cubic meter, compared to US$ 0.26 to 1.10 in tropical countries and over US$ 4.00 for small-scale producers in Latin America. With only 6-8% of global timber production entering international trade and environmentally sensitive markets only existing in Europe and North America, producers from developing countries have significantly less access to premium markets. As a consequence, timber imports from industrial countries increasingly originate from industrialised countries (Chan and Pattberg, 2008; Pattberg, 2005). In sum, certification tends to have the effect of systematically privileging northern companies in contrast to small-scale forest managers in developing countries and emerging economies.

In FSC certification, most of the time, there is no premium or final price differentiation upon timber sales, but it is observable that certified timber is more easily accepted by the market. In some cases there is a premium, for instance, in the case of FSC certification in communities in Tanzania, where certification enables the communities to earn more than US$ 19 per log, compared to a previous US$ 0.08 (FSC, 2010). Central to the Tanzanian project's success is consumer demand for sustainably harvested timber (particularly in the international market), an important driver for future community wood production in the country.

Other surveys of changes encountered in certified areas have concentrated on the economic aspects of national markets, such as studies focused on Bolivia, Malaysia and the USA. On the whole, in such countries certification has promoted better access to the market and higher prices, especially for the most processed hardwoods (Kollert and Lagan, 2005; Nebel et al., 2005; Newsom et al., 2005).

In the northern hemisphere, the FSC is perceived as more ambitious in terms of environmental and social requirements than the national or supplier driven certification systems (Cashore et al., 2005; Gulbrandsen, 2004; Keskitalo et al., 2009; Rametsteiner and Simula, 2003). The pressure for developing more ecologically integrative and socially sensitive practices is more explicit in the more externally and internationally audited FSC system. In countries like Russia, requirements for conservation are significantly higher and more explicit in the FSC, whereas in the Nordic countries and the USA the conflict between the FSC and other systems centers more on who has authority than the level of conservation.

In general, the most positive aspects found were economic and social, while the most negative refer to the certification process and its cost.

2.3 Institutional context and requirements

It was observed that some institutions influence the success of certification in terms of biodiversity conservation both in national and international certification systems. Two categories can be mentioned: (a) formal institutional requirements; (b) cultural and social requirements. The following formal requirements support certification:

- effective formal institutional infrastructure and forest legislation;
- effective laws on property or land rights;
- institutional framework or governing structure that permits distribution of benefits in case of community involvement;
- verification for certifying timber quantity and certification of local impact and, if possible, biodiversity conservation analysis.

Case studies have shown that forest certification has been most successful in states which have a conducive forest governance framework which guarantees the enforcement of forest laws; and provide land tenure security (Ebeling and Yasué, 2009; Guénéau and Tozzi, 2008). Therefore, it was found that at present, there are few developing countries where forest certification is likely to achieve widespread success (Ebeling and Yasué, 2009), Actually, this may be one reason for the fact, that currently 87.75% of FSC-certified forests are situated in the temperate and boreal zone and only 12.75% in the tropics and subtropics (FSC, 2010). Tropical countries often lack the infrastructure to facilitate certification and without the assistance of states, incentives to join a private regulatory system may be too weak (Pattberg, 2005). On the other hand, certification is not in a position to effectively compensate for the shortcomings of public action. If illegal harvesters cannot be excluded from the resource, the incentive for legal harvesters to harvest at a sustainable rate is reduced, if not entirely eliminated (Schepers, 2010). Another aspect is that a large part of the interventions in forests occurs outside the market economy or within informal economic systems. This applies for instance for fuelwood collection. A large amount of wood is produced and consumed in developing countries; a high percentage of this is used for energy consumption (FAO, 2010). Therefore, forest certification is not able to address all aspects of forest protection (Guénéau and Tozzi, 2008).

Nevertheless, certification has indirectly contributed to defining sustainable forest management standards by helping to reach an agreement on the definition of the good practices that are introduced into national legislation (Guénéau and Tozzi, 2008; TEEB, 2011).

The cultural, social and economic requirements are related to consumer maturity in including considerations on sustainability and even the level of the country's economic development in consumer decisions. Vallejo and Hauselmann (2000) point out the importance of consumer activists who can promote forest management and certification. "Consumer organisations can play an important role in initiating and advocating change in consumption patterns, and have the means to provide consumers with information that allows them to make informed choices. Consumer organisations can play a role – and have the skills to do so – in encouraging governments and industry to adopt policies and methods that will promote sustainable consumption" (Vallejo and Hauselmann, 2000: 27). If governments (e.g. German Government, 2007) lead the way in adopting national procurement policies to purchase only certified forest products for construction purposes, they could give an example and accelerate a change in consumer behaviour (Schepers, 2010).

3 The role of forest certification in a policy mix

As we have seen above, the participation of the State is important to ensuring the efficiency and effectiveness of the certification process. Conditions for a contribution of forest certification to biodiversity conservation include a conducive forest governance framework and a certain level of land tenure security in the forest country. Certification needs the coercive power of governments to clamp down on illegal trading of forestry products (Schepers, 2010). The rise of certification systems has generated new challenges and opportunities for conventional state regulation as well as interaction between private and public regulation. These have, at best, produced more credible and effective governance structures, while they run a risk of mere legitimising of existing practices (Bartley, 2010; Cashore et al., 2005; Keskitalo et al., 2009; Potoski and Prakash, 2004). At worst, certification can actively compete with state regulation, undermining both standard setting apparatus. The presence of multiple norms within a given country or region (e.g., FSC and Cerflor in Brazil) may lead to confusion on the part of consumers, while it can also undermine compliance with the more rigorous standards. On the other hand, it is not always desirable to enforce the most rigorous standards where it is important to show progress toward incorporation of a larger proportion of forests under certification norms. Adaptive and gradual adoption starting with less rigorous criteria, such as those adopted in the SLIMF system, enhance the prospects of system expansion.

Perhaps the most promising development that the certification schemes can provide for a policy mix, is the pressure to consider several different governance and control systems simultaneously and hence, to allow interaction across public and private boundaries. According to WWF (2010), certification alone cannot solve the challenges of sustainable forest management, stating: "[Certification] is a tool which works. It is time for governments and international institutions that aim to promote more sustainable management of tropical forests to make more and better use of it." (WWF, 2010: 30). It is up to all stakeholders to ensure that the tool is properly and effectively used in conjunction with other complementary tools and policies such as government regulation and consumer awareness.

Annex 1. FSC certification principles

In the case of FSC, accredited institutions adopt the ten principles that should be applied to the forest management operation. They are:

1 – Compliance with FSC Law and Principles
The forest management should respect all laws applicable in the country where it operates, international treaties, agreements signed by the country and compliance with all FSC P&C;

2 – Responsibilities and Rights of Ownership and Land Use
The rights of ownership, long-term land use and forest resources should be clearly defined, documented and legally established;

3 – Rights of Indigenous Peoples
The rights and costumes of indigenous people to own, use and manage their land, territories and resources shall be recognised and respected;

4 – Community Relations and Worker Rights
The forest management activities shall maintain or enhance the long-term social and economic welfare of forest workers and local communities;

5 – Benefits from the Forest
The forest management operations shall encourage the efficient use of multiple forestry products and services to ensure the economic feasibility and a wide range of environmental and social benefits;

6 – Environmental Impacts
Forest management shall conserve ecological diversity and its associated values, water resources, soil and fragile and singular ecosystems and landscape and thus, maintain the ecological functions and forest integrity;

7 – Management Plan
The management plan, appropriate to the proposed operational scale and intensity, shall be written, implemented and updated. The long-term objectives of forest management and the way to attain them shall be clearly defined;

8 – Monitoring and Evaluation
The monitoring shall be conducted as per scale and intensity of Forest management to assess forest conditions, forest product yields, custody chain, management activities and social and environmental impacts;

9 – Maintenance of High Value Conservation Forest
The management of high conservation value forest shall preserve or enhance the attributes which define such forests. Decisions related to high conservation value forest shall be always considered with precaution;

10 - Plantations
The plantation shall be planned and managed according to the P&C Nos. 1 to 10. Taking into account that plantations can provide a wide range of social and economic benefits and contribute to meet the requirements for global forest products, it is recommended that they contemplate management, reduce pressures and promote the restoration and conservation of natural forests.

Source: FSC.

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Towards a Framework for Assessing Instruments in Policy Mixes for Biodiversity and Ecosystem Governance

Christoph Schröter-Schlaack and Irene Ring

Summary

Whereas economic literature on instrument choice has predominantly focused on assessing single instruments, in practice most environmental problems are treated by a policy mix. Whether designed by purpose or evolved over time, policy mixes are also typical for facilitating biodiversity conservation and the sustainable management of forest ecosystem services. Building on the individual instrument reviews above, this synthesis chapter elaborates major characteristics of each instrument category as regards their roles in a policy mix and identifies theoretical interdependencies between different policy instruments. By further developing a framework for policy mix assessment this chapter provides recommendations for policy makers to explicitly consider the interaction of policy instruments when designing policy instruments for biodiversity conservation and ecosystem service management.

1 Introduction

Real-world policies aiming at biodiversity conservation and the sustainable management of forest ecosystems typically apply multiple instruments at the same time. The introduction of incentive-based approaches, such as payments for ecosystem services, ecological taxes or permit trading to facilitate conservation efforts by more traditional instruments such as protected areas or management standards has gained increasing attention over the last decades (see inter alia Bräuer et al., 2006; Drechsler and Wätzold, 2009; Hansjürgens et al., 2011b; Madsen et al., 2010; OECD, 1999). Nevertheless, economic research on instrument choice has predominantly focused on the assessment of single instrument policies. What is often missing is thus analysis of and guidance on the design of policy mixes, in particular for the field of biodiversity conservation. How to combine different instruments in order to derive better performance of the policy mix? Are there any hierarchical or sequential relationships between different measures, i.e. are some instruments enabling the use of others? Are there overlaps between different instruments that might cause inefficiency and over-regulation of policy addressees and their activities? How does one minimise transaction costs associated with different instruments being applied simultaneously?

To answer these questions it will be necessary to reconsider the characteristics of the single instruments chosen for this review. The above contributions have reviewed literature on selected instruments for biodiversity conservation and the sustainable management of forest ecosystem services regarding their: (a) effectiveness for biodiversity conservation and ecosystem service provision, (b) cost-effectiveness or other means of economic efficiency, (c) social and distributional impacts, and (d) institutional context and legal requirements. The objective of this chapter is to synthesise the main findings of the individual literature reviews and to work towards recommendations regarding potentially beneficial policy mixes for biodiversity conservation and ecosystem service management to guide the case study assessment of different policy mixes to be conducted. A first step for this synthesis is thus to summarise the insights from the individual instrument reviews (see section 2) in order to emphasise strengths and weaknesses of different instrument categories. Building on these results and Elinor Ostrom’s recent work on social-ecological systems (Ostrom, 2007, 2009; Ostrom and Cox, 2010), section 3 develops a three-step framework for assessing policy mixes and deriving recommendations regarding instrument design while explicitly considering instrument interaction. The three steps will outline the major challenges we face when designing policies for biodiversity conservation and ecosystem service management, help clarifying the role of (economic) instruments within a policy mix and lastly introduce design and evaluation criteria to maximise the value added of instruments in a policy mix. Section 4 summarises the main results and derives some conclusions.

2 Main findings of the individual instrument reviews

2.1 Overview of reviewed instruments

Besides 'direct regulation' for biodiversity conservation, including e.g. protected areas, management standards in agri- and silviculture and zoning regulation by spatial planning (see Schröter-Schlaack and Blumentrath, this report), the above instrument reviews focused on incentive-based instruments, such as payments for environmental services at national (PES) (see Porras et al., this report) as well as international level (REDD) (see Chacón-Cascante et al., this report), ecological fiscal transfers (see Ring et al., this report), tax reliefs (see Oosterhuis, this report), forest certification (see Kaechele et al., this report) and the extension of offsets in the form of mitigation and habitat banking (see Santos et al., this report)[1]. Economic theory on instrument choice deems 'incentive-based instruments' to be more flexible and cost-effective substitutes to traditional 'direct regulation' policies. However, as was shown in the introduction of this review, mixing different policies for biodiversity conservation – intended or unintended – is not only a matter of fact in political reality but justifiable from various non-economic perspectives (see Ring and Schröter-Schlaack, this report) and in some circumstances even desirable from an economic standpoint. Hence, the task is not (only) about substituting one instrument by another but to derive recommendations on how to combine them in a way beneficial for biodiversity conservation and sustainable management of forest ecosystem services.

The basic idea behind designing policy mixes is to overcome weaknesses of single instrument policies, such as low ecological effectiveness, high abatement costs (including opportunity and transaction costs) of environmental goal attainment, unjust distribution of environmental burdens or abatement costs among the affected stakeholders or (prohibitively) high transaction costs. In order to assess the incremental contribution of incentive-based instruments to the existing mix of local, national and international policies for biodiversity conservation and ecosystem service provision it is necessary to reconsider the main characteristics of the available solutions. Hence, table 1 summarises the main findings of the literature reviews regarding the four criteria of assessment and turns them into hypotheses regarding the performance of the considered instruments. These hypotheses will be one starting point for the research questions of the case studies to be conducted later within the POLICYMIX project. Table 1 clusters the findings as follows:

- Environmental effectiveness, i.e. was the environmental goal reached by the use of the instrument (e.g. preservation of natural landscape, habitat protection or species conservation)?
- Cost-effectiveness, i.e. was the environmental goal reached by the lowest costs? Besides opportunity costs this also comprises implementation and transaction costs associated with the specific instrument.

[1] *Subsidies in sectors other than biodiversity policies can often act as adverse incentives promoting environmentally harmful activities. If relevant, harmful subsidies will be covered in terms of the institutional context in the POLICYMIX case studies.*

- Social and distributional impacts i.e. are there any positive or negative social impacts associated with the use of the instrument and how are benefits and cost distributed among social actors? Although social impacts cover more than just distribution (e.g. fairness and legitimacy of decision-making processes as well as participation) most published work considered for the reviews above has focused on distributional aspects of different instruments.
- Institutional arrangements required, i.e. which institutions are necessary for successful implementation and operation of the instrument?

As can be seen from Table 1 the toolbox for policy makers to answer the biodiversity challenge is well equipped. Although all reviewed instruments aim at conserving biodiversity and sustaining (forest) ecosystem service provision they do so by very different mechanisms. On a general basis, one could distinguish between 'direct regulation', incentive-based approaches and market facilitation. Whereas 'direct regulation' operates by either direct public provision of biodiversity conservation (e.g. protected area designation) or standard setting (management or pollution standards, spatial planning), incentive-based instruments do so by providing financial (dis-)incentives to stakeholders. Within the group of incentive-based instruments one could further distinguish between price-based mechanisms and quantity-based approaches. The former comprise tax reliefs, payments for environmental service provision or biodiversity conservation to different actors at different governance levels (PES, REDD, ecological fiscal transfers) while tradable permits are counted to the latter. Finally, there is a third category: informative and motivational measures provide knowledge to actors about the consequences of their behaviour, thereby facilitating intrinsic motivation for self-regulation in conserving biodiversity or managing ecosystem services. Figure 1 below places these three policy instrument categories in a continuum stretching from direct government allocation of land and resources for conservation (far left) to more indirect interventions aiming at correcting allocation failures of existing markets (far right).

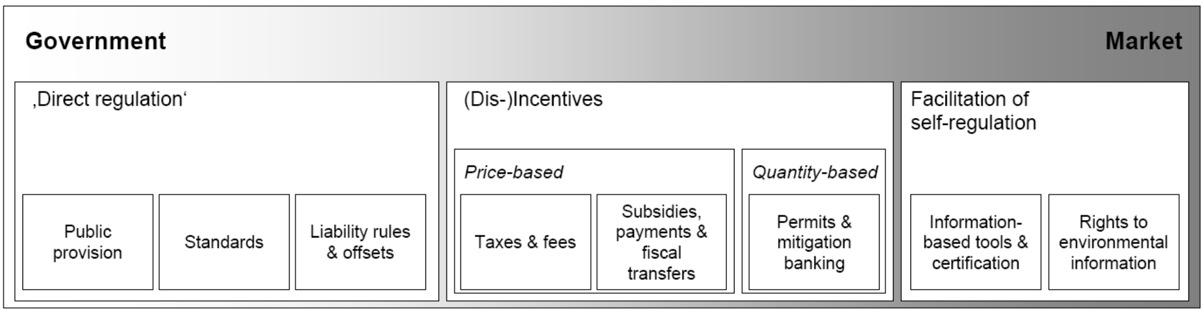

Figure 1. Continuum of policy instruments for biodiversity conservation
Source: own representation

The following sections shortly summarise the main findings of the literature reviews regarding the selected instruments covered in these categories.

Table 1. Hypotheses on performance of selected single instruments for biodiversity conservation

Instrument type	**'Direct regulation', e.g. Protected Area (PA) designation**	**Offsets, Habitat Banking and Permit Trading**	**Tax reliefs**	**Ecological fiscal transfers**	**Reducing Emissions from Deforestation and Degradation (REDD and REDD+)**	**Payments for Environmental Services (PES)**	**Forest certification**
Goal	Safeguard important areas for species and habitat conservation	Account for and mitigate inevitable impacts on biodiversity and ecosystems	Account for positive environmental externalities provided by land users	Compensating decentralised governments for opportunity and / or management costs as well as spillover benefits of protected areas (PA)	Multinational and multilevel policies and measures to reduce deforestation and forest degradation and associated carbon emissions in develop-ing countries (REDD), while considering conservation and co-benefits (REDD+)	Incentivising land users for biodiversity conservation and ecosystem service provision, e.g. by compensating for associated opportunity and management costs	Promote biodiversity- and environmental-friendly forest production in accordance with legal codes and certification requirements
Actors addressed	Private and public actors	Private and public actors	Private actors	Public actors	Public and private actors	Mostly private actors / land users	Private actors (consumers)
Baseline and policy context	Protection provided by other primary in-struments (e.g. emission / manage-ment standards) or existing PA network, very often no protection at all	Impacts allowed by (management / emission / performance) standards	Tax payers behaviour without the tax relief (business as usual might be biodiversity friendly anyhow)	PA coverage when instrument is introduced	Deforestation and degradation rates without REDD (i.e. business as usual de-fined e.g. by historical or forecasted defor-estation rates or na-tional circumstances)	Land-use practice without incentives by PES schemes (business as usual could be either static, declining or improving)	National forestry regulation, certification process most often progressive and adaptive
Conservation effectiveness	High – increase in / conservation of biodiversity and ecosystem service provision; however, effectiveness may be at risk due to weak enforcement or may erode in the future due to changing environmental conditions (e.g. climate change)	Medium – although typically designed to allow for a "no net loss"-goal, problems arise in assuring equivalence of mitigation measures and their long-term monitoring	Low – depending on tax burden relieved (existence of tax, actual enforcement of payments, and sufficient tax rate); non-targeted approach	Medium to high – increase in quantity and quality of PAs likely (especially when beneficiary of transfers can influence quantity and quality of PAs)	Potentially medium to high – depending on actual design (additionality, avoidance of leakage, permanence and carbon accountability) of scheme once established	Low to high – depending on instrument design regarding baseline, and additionality, leakage, permanence and participation	Medium – impacts dependent on rigorousness of standard and framing conditions, such as intensity of investment, difficulties in transport and licensing, land tenure and conflicts with competing land uses

Table 1 (cont.). Hypotheses on performance of selected single instruments for biodiversity conservation

Instrument type	**'Direct regulation', e.g. Protected Area (PA) designation**	**Offsets, Habitat Banking and Permit Trading**	**Tax reliefs**	**Ecological fiscal transfers**	**Reducing Emissions from Deforestation and Degradation (REDD and REDD+)**	**Payments for Environmental Services (PES)**	**Forest certification**
Associated costs and proxies for cost-effectiveness	Medium – though PAs very often show a positive benefit-cost-relationship, local opportunity costs can be substantial	High – in particular the option to trade mitigation measures significantly reduces opportunity costs; however, some ecosystem / habitat types may be (to) costly to restore	Medium – low transaction costs as resting on existing administrative procedure; however, very often incentives provided insufficient for required change in land-use practice	Medium to low – low transaction costs as it builds on existing mechanism (fiscal transfer schemes and PA designation)	Potentially medium to high – pilot schemes may have underestimated implementation and transaction costs of fully developed REDD architecture	Medium to high – no up-front public investment for buying land, auction-based programmes limit excessive rents; however potentially high transaction costs	Medium – administrative costs of certification scheme may be substantial (in particular in tropical forests)
Social impacts	Medium – ecosystem services protected by PAs may benefit (local) population; however, substantial opportunity costs and risk to revoke informal rights (e.g. access / abstraction) in area designation	Medium – increase in education / job and income opportunities for rural landowners marketing offsets; compensation of opportunity cost of land conservation (TDR)	Medium – compensation for opportunity costs of environmentally friendly land-use practices; however, only applicable to tax debtors (e.g. landowners)	Medium – depending on entry point of PAs in fiscal transfer systems; fiscal transfers as such address inequalities between jurisdictions	Potentially high – depending on the institutional infrastructure at international and national level to enable broad participation of and within developing countries	Medium – support of rural livelihoods, resource management and social coordination capacities; but enrolment numbers limited by insecure property rights and transaction costs, mixed effect on poverty alleviation	Low to medium – difficult to reach smaller operators except through subsidised schemes; communities are often benefited through workforce participation and engagement in co-benefits
Legal and institutional requirements	Medium to High – easily introducible for a few unique spots; increasingly difficult to implement if demand for land is highly competitive	High – strong public sector involvement necessary in standard setting and monitoring of mitigation measures, high up-front investment for trading architecture	Low – tax deductions are likely to be politically accepted; implementation builds on existing administrative structures	Medium – requires existing fiscal equalisation scheme; introduction of PA indicator often needs constitutional changes and new laws, requiring political majorities	Medium to high – countries need to be able to participate internationally / implement national and sub-national levels programmes; this may require broad stakeholder participation, reform of national forest laws and creation of new institutions	Medium to high – definition and enforcement of property rights key for programme success, more effective programmes require high up-front costs for baseline setting, negotiations, fund- and awareness raising	Medium to high – effective forest legislation / laws on property rights; architecture to distribute benefits in case of community involvement

2.2 'Direct regulation'

Public provision of biodiversity conservation and standard setting

'Direct regulation' – covering a broad range of measures – is the most widely used approach for environmental protection and this holds true for biodiversity conservation, e.g. in the form of protected areas, management or pollution standards. Typically such measures are perceived as highly effective but costly (mainly in terms of abatement costs) tools for biodiversity conservation. 'Direct regulation' is able to safeguard a safe minimum standard of conservation, making it an important ingredient to any conservation strategy. Its social and distributional impacts are somewhat mixed. On the one hand, 'direct regulation' clarifies property rights and thus makes use or access rights legally enforceable. This is an important enabling condition for the use of market-based instruments for conservation and ecosystem management that require clearly defined property rights to work effectively. There is however the risk to preclude informal property rights, e.g. those of indigenous people. As 'direct regulation' tends to ignore differences in opportunity costs of conservation across actors, there is a distinct operating space for incentive-based approaches to complement these instruments. See Schröter-Schlaack and Blumentrath (this report) for more details.

Offsets, mitigation and habitat banking

Biodiversity offsets are defined as measurable conservation outcomes resulting from actions designed to compensate for residual biodiversity impacts from project development. They hence allow deviating from a performance or pollution standard, if – after appropriate prevention and mitigation measures have been taken – associated impacts are offset. Biodiversity offsets are part of the legal framework in several countries and essentially local and bioregional tools tailored to local circumstances and usually planned within the same bioregion as the area impacted. Mitigation or habitat banking can be seen as an extension of biodiversity offsets, turning offsets into assets that can be traded, and thereby allowing for a cost-effective production of offset measures (see also permit trading below). However, habitat banking faces some theoretical and implementation problems in aiming at the desired biodiversity outcomes. These are mainly related to equivalence and additionality issues, which could lead to unintended economic costs and environmental consequences. See for more details Santos et al. (this report).

2.3 Incentive-based approaches

The main difference between 'direct regulation' and economic instruments is that only the latter operate by financial incentives. They are based on the assumption that private (and in the case of fiscal transfers and REDD also public) decisions on biodiversity conservation are primarily taken on the basis of financial cost-benefit considerations (see Oosterhuis, this report). Hence, incentive-based instruments strive to alter private costs and benefits so that any unaccounted social costs (and benefits) of environmental degradation can be 'internalised' to ensure the desired environmental improvement (Barbier et al., 1994: 179). They can do so by either providing positive incentives (subsidies, tax reliefs, fiscal transfers or payments) for providers of biodiversity conservation and ecosystem services, or by burdening biodiversity-harmful activities and (excessive) use of ecosystem services (environmental taxes, necessity to hold a permit, obligation to buy offsets).

Payments for environmental services

Payments for environmental services (PES), also known as payments for ecosystem services, are effective in providing incentives to stimulate forest protection and improved silvi- and agricultural practices that sustain ecosystem service provision. They may also help to align local malpractices through negotiations between parties where no legislation exists. Whereas in Latin America PES is a well-known and widely used approach, it is only starting to be developed in Southeast Asia and Africa. In Europe government-financed schemes dominate, the main example being agri-environmental schemes co-financed by the EU's Common Agricultural Policy. Although little is known empirically about the cost-effectiveness of existing PES schemes, start-up and transaction costs of such programmes tend to be high. Evidence suggests that PES programmes have at best a mixed effect on poverty – in practice they tend to benefit wealthier, better connected and informed landowners in possession of better assets and larger properties. See Porras et al. (this report) for more details.

REDD and REDD+
REDD can be understood as an architecture of multinational and multilevel policies and measures to reduce deforestation and forest degradation. As it is still being negotiated its effects may yet only be projected. If directed at reducing carbon emissions and providing biodiversity conservation and other 'co-benefits' it sometimes is referred to as REDD-plus. In a narrow and mechanistic sense REDD (REDD-plus) can be seen as a voluntary mechanism that entitles developing countries (and communities and individuals as well) monetary compensation for their reduction in carbon emissions through avoided deforestation and forest degradation, one source of global carbon emissions. REDD will be a multinational scheme and also a multilevel approach as beneficiary countries must foster a national payment mechanism to compensate land users and communities for their efforts. REDD actions are to be based on monitoring, reporting and verification, and require a nested governance structure stretching over different government levels. Ultimately, the success of REDD will depend on the capacity of the host countries to implement their national programmes properly. As such REDD is much more an architecture than a single instrument policy. See Chacón-Cascante et al. (this report) for more details.

Tax reliefs
Tax reliefs can be an attractive tool for biodiversity conservation because they use to a large extent the existing tax system and infrastructure for administration, assessment, payment, monitoring and enforcement, thereby minimising transaction costs for instrument implementation. However, its ecological effectiveness may also be low or uncertain, since tax reduction can only be applied if a tax exists in the first place that is actually enforced and if the tax rate is sufficiently high so that a relief induces actual change in addressee's behaviour. Moreover, tax reliefs do not allow for a targeted approach. See Oosterhuis (this report) for more details.

Fiscal transfers
Fiscal transfers have long been used to support local governments in providing public goods, though seldom transfers are assigned on indicators representing biodiversity conservation efforts, like done in Brazil and very recently implemented in Portugal. Although introduced as compensation to public actors at various governmental levels for opportunity costs associated with land-use restrictions imposed by protected areas, fiscal transfers in Brazil have developed as an incentive to designate more protected areas. Conservation effectiveness of the tool will be higher, if transfers are assigned not only on quantity but also on quality and additionality of protected areas. However, this requires reliable monitoring which will increase the transaction cost of the approach, if PA monitoring is not yet executed by conservation authorities. See Ring et al. (this report) for more details.

Permit trading
In contrast to the price-based approaches sketched above, permit trading operates by fixing a maximum quantity of pollution or ecosystem service use. Rights to pollute or use a certain ecosystem service are certified, and certified rights are tradable among users. The permit price – and thus the financial disincentive to ecosystem service occupation – is determined by demand and supply on the permit market. Trading schemes can be applied for biodiversity offsets (see above) or for reducing environmental pollution or to safeguard natural landscapes from excessive land development and urban sprawl. By fixing the maximum environmental load tradable permit schemes are able to safeguard a minimum standard

and are hence an increasingly considered tool for biodiversity conservation. They may realise cost-effective allocation of compliance measures, thereby minimising opportunity costs of reaching conservation goals. However, permit trading needs a clear regulatory framework to reach conservation gains, especially since spatial allocation of conservation efforts or resource use can only be realised at high transaction costs, easily exceeding potential gains from trading. See Santos et al. (this report) for more details.

2.4 Information-based approaches

Informative approaches operate by providing policy addressees (additional) information about the impacts of their activities regarding biodiversity and ecosystem services, irrespective of productive or consumptive utilisation. Moreover, 'Right to Environmental Information'-acts are being used by NGOs to gain access to information on biodiversity data from land owners (or authorities). Only recently, the European Court of Justice (ECJ) ruled that NGOs in Germany have the right to challenge in court projects that may have a significant impact on protected areas (ENDS Europe, 2011).

Informative approaches are based on the assumption that actors will reconsider their decisions once a more comprehensive picture of all consequences is drawn. While some information campaigns are targeted at non-use values, e.g. existence or bequest values, other informative approaches are targeted towards marketable ecosystem services and the promotion of biodiversity-friendly management and production. Lastly, information may be provided to different audiences by different approaches: eco-labeling addresses consumers, while certification also addresses companies looking for inputs to production as well as wholesale purchasers.

Forest certification
Forest certification acts as a bridge between market regulation and environmental governance by furnishing specific criteria in response to consumer and buyers' demands for sustainably produced timber. Certification acts as a non-state arbiter of conformity with quality and performance criteria in achieving environmental goals. Incentives for producers to certify their stands consist of potential price premiums certified timber and non-timber forest products may generate on the market. Evidence suggests however, that these incentives are often insufficient to prevent deforestation, although demand for certified products is growing globally. See Kaechele et al. (this report) for more details.

3 Assessing instruments in policy mixes for biodiversity conservation and management of ecosystem services

3.1 A necessary caveat at the outset

In a mechanistic view, the objective of biodiversity conservation policies may be described as to achieve the level of biodiversity conservation that meets the needs of society (Stoneham et al., 2003) now and in the future. However, as could be seen from the introductory part of this review, biodiversity is an inherently complex and dynamically evolving system (OECD, 1999); rather a scientific concept than a manageable objective in itself. Each ecosystem – and hence the services it provides – has very different ecological characteristics, is accessed and affected by different groups (in size and composition) of stakeholders, threatened by different pressures and is at a different stage of disturbance and hence proximity to a threshold of irreversible change. Even within forest ecosystems, depending on geographical scale and location, a myriad of different sub-ecosystems may be identified. Thus, it seems to be impossible to derive some general conclusions on the design of policy instruments or their potentially beneficial combination without ignoring the necessary level of detail of each specific ecosystem and its societal embedding or oversimplifying either the complex biophysical interplay of the different components subsumed under the term biodiversity or the institutional surrounding shaping its management and use.

What is more, as Gunningham and Sinclair (1999: 51 ff.) have outlined, the many possible permutations of instruments and institutional interactions as well as the influence of localised political and cultural traits on instrument acceptance and performance make the task of producing general conclusions and recommendations on how to combine different instruments even more challenging. Starting from a theoretical textbook standpoint it seems to be easy to distinguish different instrument categories, their working mechanisms and predicted performance. In positive analysis, however, clear categorisation and definition of instruments and their modus operandi become indistinct. Real-world policy comprises manifold design options that sometimes have tremendous influence on its performance. Thus, the more detailed the analysis gets, the less clear cut are its results. Another challenge results from the fact that politicians will very often not be able ‘to design’ optimal policy mixes. The policy context will rather limit their degrees of freedom and thus the set of available and appropriate instrument options. Hence, many policy mixes are the result of limiting factors and introducing new instruments will be sometimes the only possible response to counter an insufficient (but persisting) existing instrument.

3.2 A three-step framework to assess instruments in policy mixes

Notwithstanding these methodological challenges, we think it to be worthwhile to work towards the development of a framework and recommendations for designing policy mixes for biodiversity conservation and sustainable management of forest ecosystem services. This will be based on the specific characteristics of biodiversity and forest ecosystems on the one hand and the frameworks for policy mix assessment and the classifications of instrument interaction presented by Ring and Schröter-Schlaack (this report) above on the other. Building on this background it will be possible to derive recommendations on how to equip the policy response to the ongoing loss of biodiversity and ecosystem degradation.

The remainder of this section thus introduces a three-step framework for assessing policy mixes and derives hypotheses regarding their design to be tested in the case studies of the POLICYMIX project (see Table 2). The framework’s three fundamental steps for designing policy responses are built up by several coarse grain assessment categories. These broad assessment categories can be further subdivided into relevant issues to consider in steps 1 and 2, and into fine grain assessment criteria for the detailed evaluation and design of policy instruments in step 3. We choose these tiers in analogy to Elinor Ostrom’s institutional analysis and development framework (Ostrom, 1990; Ostrom et al., 1994) and her work on analysing social-ecological-systems (SES) (Ostrom, 2007, 2009; Ostrom and Cox, 2010) to cope with the trade-off between a necessarily detailed level of empirical analysis and more general conclusions and recommendations toward policy development and instrument design. Moreover, it is possible to frame some of the analytical steps proposed below in a driver-pressure-state-impact-response framework (DPSIR) (Gabrielsen and Bosch, 2003; Smeets and Weterings, 1999), or by its alterations developed in the literature (Maxim et al., 2009; Rudd, 2004). Although we explicitly refer to the structure of the DPSIR framework and Ostrom’s SES-analysis, the framework to be developed for our purposes requires a special focus on assessing policy instruments and policy mixes against policy objectives regarding biodiversity conservation and ecosystem service management in order to derive recommendations for policy (mix) design – an aspect only brushed by the SES- and DPSIR-frameworks.

Table 2. A three-step framework for assessing and designing policy mixes for biodiversity conservation and ecosystem service management

First Step	Assessment category	Issues to consider
Identifying challenges and context Scoping phase	Characteristics of biodiversity and ecosystem services	Potential trade-offs between biodiversity and ecosystem services
		Irreversibility of biodiversity loss
		Tipping points and threshold effects
		Lacking property rights for biodiversity and many ecosystem services
		Defining ecosystem service in question
	Objectives regarding biodiversity conservation and ecosystem service management	Range of ecosystem services utilisation
		Trade offs between different ecosystem services
	Drivers of biodiversity loss and ecosystem degradation	Direct and indirect drivers from various sources / better “sectors” than sources?
		Negative impact of drivers amplified by sectoral policies
	Actors and governance levels	Public and private actors
		Local to global level actors
		Alteration of decision-making processes and inputs across scales – and thus necessary policies
	Cultural and constitutional settings	Local knowledge and traditional practices
		Relative appropriateness of monetary valuation and market-based conservation in cultural context
		Constitutional options and constraints

Second Step	Assessment category	Issues to consider
Identifying gaps and choosing instruments for analysis Evaluating the functional role of instruments in the policy mix	Policies in place versus new instruments under consideration	Policy mix across sectors and governmental levels (national/federal versus regional/local)
		Experience with policy instruments
		Persistence of existing instruments
	Context-specific strengths and weaknesses of instruments	Dealing with uncertainty and ignorance
		Lacking property rights
		Spatial targeting of instrument
		Additionality
		Type of ecosystem service
	Instrument interactions	Inherently complementary interaction
		Inherently negative interaction
		Sequencing/path-dependency
		Context-dependent interaction

Third Step	Assessment category	Assessment criteria
Policy evaluation and design Impact evaluation for existing (ex post) and scenario analysis for new instruments (ex ante)	Conservation effectiveness	See WP3 guidelines
	Cost-effectiveness and benefits	See WP4 guidelines
	Distributive impacts and legitimacy	See WP5 guidelines
	Institutional options and constraints	See WP6 guidelines

The first step comprises the identification of the context and the main challenges for a policy response aiming at biodiversity conservation and ecosystem service management. These challenges establish fundamental issues to consider for the choice of suitable instruments, the assessment of the functional roles of different instruments in the relevant policy mix, and later on the assessment criteria for the evaluation and design of the instruments in this policy mix (section 3.3). The second step includes criteria and recommendations regarding the choice of instruments, about the functional role different instruments might play in addressing the challenges highlighted in step 1 and how interactions between instruments in policy mixes could be considered. For this purpose, we build on the classification of Gunningham and Sinclair (1999) and its advancement by Flanagan et al. (2010) as introduced earlier (Ring and Schröter-Schlaack, this report), while adapting our framework to biodiversity conservation and forest ecosystem governance regarding the relationship between instruments and the role of individual instruments within a policy mix (section 3.4). Lastly, the third step elaborates specific design issues in order to maximise the value added of single instruments within policy mixes for biodiversity conservation and ecosystem service management (section 3.5).

3.3 First Step: Identifying challenges and context

When it comes to analysing policy mixes, the focus is not on maximising effectiveness or efficiency of individual policy measures but on the complementarity of the instruments involved, their interplay and the ability of the policy mix to address all drivers of the underlying problem (see Ring and Schröter-Schlaack, this report). The appropriate mix of instruments and actors will hence depend upon the nature of the environmental problem, the target groups and wider contextual factors (see Gunningham et al., 1998).

Against this backdrop, the first step of the proposed framework consists in gaining a thorough understanding of the policy object, i.e. biodiversity conservation and (forest) ecosystem services management. Although we believe the questions listed in Box 1 to be neither comprehensive nor exclusive, they may cover the most relevant questions to answer. Thus the following subsections strive towards answering these questions and identifying the challenges and context associated with biodiversity conservation and ecosystem service management.

Box 1. Challenges and context for policy responses aimed at biodiversity conservation and ecosystem service management

Firstly, what are important characteristics of biodiversity and ecosystems that will influence appropriateness, applicability and success of certain instruments and their combinations?

Secondly, what are the policy objectives regarding biodiversity conservation and ecosystem service management?

Thirdly, what are the drivers of biodiversity loss and ecosystem degradation and how might these be adequately addressed?

Fourthly, who are the main actors whose behaviour is impacting biodiversity and influencing ecosystem service provision and how do these actors derive their decisions?

And lastly, what are cultural and constitutional constraints (or enabling conditions) that may hamper (or facilitate) the inclusion of certain policy instruments?

3.3.1 Addressing characteristics of biodiversity and ecosystem services

When striving for recommendations on how to design policies (or policy mixes) for biodiversity conservation and sustaining ecosystem service provision, it is imperative to carefully distinguish between the two and to identify the inherent trade-offs. It is still largely unclear, how biodiversity contributes to ecosystem functioning and thus the sustained provision of ecosystem services (e.g. Balvanera et al., 2006; Bengtsson, 1998; Luck et al., 2003; Schwartz et al., 2000; Swift et al., 2004), and, in turn, what effects aligning nature conservation and ecosystem management towards ecosystem service provision will have on biodiversity (e.g. Díaz et al., 2011; Norgaard, 2010). While an ecosystem service approach can help to recognise values and guide management of ecosystems, it does not explain how ecosystems function (TEEB, 2010a). It is likely that a focus on ecosystem service provision will overemphasise some components of biodiversity, e.g. certain habitats or species that are deemed to be important for certain ecosystem services. Other components, like those still unknown or those whose functions are not (fully) understand yet, may slip out of conservation priorities, thereby threatening the long-term functionality of ecosystems and their resilience. Hence, a precautionary approach to conserving biodiversity is advisable to maintain resilient ecosystems, capable of delivering multiple services (Elmqvist et al., 2010). In the following, we briefly describe the main challenges associated with biodiversity conservation on the one hand and sustaining ecosystem service provision on the other.

Biodiversity

Following the Millennium Ecosystem Assessment (2005a: 18), biodiversity is defined as "the variability among living organisms from all sources including, inter alia, terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are part; this includes diversity within species, between species and of ecosystems." Due to the multi-dimensionality, complexity and dynamics of biodiversity, its conservation and sustainable use poses challenges to policy-makers which differ from those in other areas of environmental policy (OECD, 1999).

Firstly, loss of biodiversity and the degradation of ecosystems are in many cases irreversible (see e.g. Barbier et al., 1994: 18; Millennium Ecosystem Assessment, 2005a: 2). Although knowledge and methods on ecosystem restoration have developed in recent years (see inter alia Neßhöver et al., 2011), such restoration typically involves major costs and time lags and even in case of successful restoration many features of biodiversity will be lost forever.

Secondly, biodiversity loss and ecosystem degradation are characterised by threshold effects. There is an abundance of ecological literature available on this topic. Exemplarily, threshold effects have been defined by Muradian (2001) as sudden modifications of a given system property, resulting from the soft and continuous variation of an independent variable. Thus, they are invalidating predictions based on models and relationships that apply at lower levels (TEEB, 2010b: XXXVIII). Dupraz et al. (2009) list a range of examples for such threshold effects, such as the increase of the vulnerability to additional perturbations for ecosystems that have been previously submitted to strong anthropogenic pressure (Levin et al., 1998), modifications in the equilibrium of temperate lakes (Weisner et al., 1997), colonisation by undesired species (Asner and Vitousek, 2005) or habitat fragmentation and disappearance of species (Kennedy et al., 2002). Thus, species diversity of a landscape may decline steadily with increasing habitat degradation to a certain point, then fall sharply after a critical level of degradation is reached (TEEB, 2010b). Hence, keeping impacts on ecosystems on a 'safe' level is necessary to avoid the tremendous costs associated with crossing thresholds and ecosystem collapse.

Thirdly, our knowledge on ecological processes and their interplay constituting ecosystem functioning, our understanding of the thresholds and tipping points beyond which the provision of ecosystem services is insufficient, and lastly our knowledge on ecosystem resilience are all characterised by uncertainty and ignorance (see inter alia Chapin III et al., 2000; Elmqvist et al., 2010; Folke et al., 2004; Purvis and Hector, 2000). Thus, gaining solid ground for decisions strictly adhering to a ‘safe’ or ‘appropriate’ level of utilisation is extremely challenging. Biodiversity policies must therefore follow the precautionary principle and strive to safeguard a minimum standard of conservation that is based upon our best estimates about long-term functionality.

Lastly, benefits of biodiversity conservation are often public goods, received in spatially and temporally highly diverse patterns. Most benefits are indeed only realised in the long term. Very often property rights for species or habitats in biodiversity hot spots are lacking or if existent cannot be enforced properly.

Ecosystem services

While the arguments to support biodiversity conservation hitherto relied on its intrinsic, use and non-use values, the Millennium Ecosystem Assessment (MA) emphasised the importance of biodiversity as a source of ecosystem services (Ninan, 2007: 1), which are defined as “services humans derive from ecosystems” (Millennium Ecosystem Assessment, 2003: 53). The MA distinguished between four different categories of ecosystem services (Millennium Ecosystem Assessment, 2005b). Provisioning services are in many cases marketable goods, such as food, fibre or timber, though market failures and externalities often distort the rates of their utilisation. In comparison, regulating services, like flood control or climate regulation, and cultural services, like recreation, spiritual and aesthetical values, are often non-marketable goods that lack working institutions to demonstrate or even capture the value associated with these services. Supporting services, like soil formation, photosynthesis and nutrient cycling are core processes of functioning ecosystems and contribute only indirectly, i.e. via the other categories of ecosystem services, to human well-being.

Although the promotion of the ecosystem service concept by the MA inspired much thinking on the environment-human well-being interface, it has yet not succeeded in aligning policies towards sustaining ecosystem service provision (see inter alia Daily et al., 2009; Naidoo et al., 2008). Partially, this may be attributed to the fact that the concept of ecosystem services is somehow diffuse and vague, with many alterations in the definition of services and their classification, in particular if applied for different purposes (see inter alia Boyd and Banzhaf, 2007; Costanza, 2008; Daily, 1997; Elmqvist et al., 2010; Fisher and Turner, 2008; Fisher et al., 2009; Lamarque et al., in press). Another major difficulty is that many ecosystem services are (mixed) public goods whose use level can hardly be regulated due to information problems and institutional failures (de Groot et al., 2010: 12). The latter refer to unintended incentives provided by markets (externalities) or governments (e.g. environmentally harmful subsidies); the former to the poor knowledge on the contribution of ecosystem services to human welfare and how human actions lead to environmental change with impacts on human welfare. Ultimately, ecosystem services are a cross-cutting concept as environmental protection as such is. Many aspects of ecosystem services are already regulated by a range of sectoral policies, thus intense policy coordination is required.

Another challenge to facilitate a sustainable provision of ecosystem services by policy interventions results from the differences in the spatial configuration of the benefit flows by different services. Following a classification developed by Balmford et al. (2008: 17) one could distinguish five general categories that would demand different or at least differently designed) policy instruments:

- Locally produced benefits: when the point of service production is the same as the point of use (e.g., soil production);
- Omni-directional neighbourhood benefits: when service use takes place within a buffer area surrounding the point of production (e.g., pollination);

- Directional neighbourhood benefits: when service use takes place in the neighbourhood of the area of production, but only in a given direction (e.g., storm protection);
- Long-distance directional benefits: when service users are located far from the point of production, with services flowing in specific directions (e.g., water provisioning through water flowing downstream); and
- Globally-distributed benefits: when the service can be used anywhere irrespective of the point of production (e.g. climate change mitigation by carbon sequestration).

3.3.2 Addressing multiple objectives

Biodiversity and ecosystem services affect and are impacted by many human activities and benefits, and are thus subject to a wide array of objectives and utilisation strategies from different groups within society. In particular for managing marketable ecosystem services there are strong interests from private and public actors to sustain their provision. On the contrary, biodiversity conservation and management of non-marketable ecosystem services are due to their public good characteristics first and foremost public responsibilities (TEEB, 2009: 31).

In order to understand the multiple objectives of biodiversity conservation and ecosystem management it is indispensable to become clear about the trade-offs involved. Besides the trade-off between biodiversity conservation and the management of ecosystem service mentioned above (see subsection 3.3.1), there are also trade-offs in utilising the same ecosystem service and between different ecosystem services derived from the same ecosystem.

Firstly, provisioning ecosystem services may be utilised in very different ways. Exemplarily, food may be produced with high inputs of irrigation water, fertilisers and pesticides or it may be produced in an extensive, biodiversity-friendly way. Furthermore, it is important to consider the current state of the respective ecosystem, the threshold at which it fails to deliver ecosystem services, its targeted conservation state and the best estimates of the uncertainties attached to the functioning of the ecological system (TEEB, 2008).

Secondly, trade-offs may arise between different ecosystem services provided by the same ecosystem or by ecosystems that are somehow connected. As ecosystems provide multiple services and interact in complex ways, the delivery of ecosystem services will vary in a correlated manner. Hence, if ecosystems are managed for the delivery of a single service (e.g. food production), other services (e.g. regulation services such as flood protection) are nearly always affected negatively (Elmqvist et al., 2010).

3.3.3 Addressing multiple drivers of loss

Biodiversity is exposed to a wide array of direct and indirect drivers of loss. The MA has identified five important direct drivers of change in ecosystems, namely habitat change (land-use change and physical modification of rivers or water withdrawal from rivers), overexploitation, invasive alien species, pollution, and climate change (Millennium Ecosystem Assessment, 2005b: 14ff.). Most of these direct drivers are projected to remain at least constant or are expected to grow in intensity in the future. Similarly, Sala et al. (2000) found five major direct drivers of biodiversity change (land use change, increase in atmospheric CO_2, nitrogen deposition, climate change, and biotic exchange) for the principal terrestrial biomes of the Earth. For terrestrial ecosystems land-use change resulting in habitat loss and fragmentation is found to be the predominant direct driver of biodiversity loss (see inter alia Kålås et al., 2010; Rockstrom et al., 2009; Sala et al., 2000). Moreover, direct drivers are often synergetic, i.e. land-use change can result in greater nutrient loading (if the land is converted to high-intensity agriculture), increased emissions of greenhouse gases (if forest is cleared), and increased numbers of invasive species (due to the disturbed habitat) (Millennium Ecosystem Assessment, 2005b: 14).

Indirect drivers of biodiversity loss and unsustainable utilisation of ecosystem services derive inter alia from the global population growth (UN DESA, 2009) and the associated growing demand for food, energy and raw materials, from changing diet patterns e.g. a growing demand for meat (Speedy, 2003), as well as from political decisions, e.g. on strategies for carbon neutral energy production (biofuels) to meet growing energy demand (OECD-FAO, 2009: 147 ff.).

These direct and indirect drivers originate from multiple sectors across the economy and are thus influenced by already existing policies, e.g. to regulate silvi- and agriculture, fisheries, energy production, transport, trade or resource extraction. Until now, many of these policies additionally spur the pressure on biodiversity, e.g. subsidies for agriculture, fishing fleets, energy or fuels that are not considering the negative impacts on biodiversity and the sustained capacity to provide ecosystem services (OECD, 2005; The World Bank & FAO, 2009; Valsecchi et al., 2009).

3.3.4 Addressing multiple actors and governance levels

The multidimensionality of biodiversity also implies that multiple actors (public and private) at all governance levels (from international and national to regional and local level) are to varying degrees dependant on ecosystem services and are in turn impacting biodiversity and ecosystem service provision. From a policy viewpoint it is obvious that only few instruments are able to be all inclusive, i.e. to include and incentivise all relevant actors at all levels, without becoming too vast and complex. Spoken in economic terms, the transaction costs associated with designing and implementing an instrument to cover all actors will be prohibitively high. On the other hand, very simple measures potentially capable of encompassing all relevant actors, e.g. broad information campaigns, will have only limited effect on actual behaviour and thus limited effect on biodiversity loss and ecosystem degradation.

Moreover, as different actors are working with different procedures of decision-making it is necessary to feed in the value of biodiversity and ecosystem services in different ways into decision-making. Whereas consumers might respond to eco-labelling of biodiversity-friendly manufactured products in their purchasing decisions, businesses might adapt to reformed liability rules for their products or react to taxes on environmentally harmful inputs to their production chain. Governments may respond to the possibility to sell carbon (and biodiversity) credits when actively reducing deforestation rates or the incentive to receive fiscal transfers if they designate protected areas within their territory.

Lastly, decision-making and thus policies to influence decision-making have to be altered across scales. Very often decision-making authority rests at different governance levels, i.e. the scope of local decisions is shaped by regional decrees, whereas regional decisions are influenced by national and / or international standards and prescriptions. However, it may also be the other way round: biodiversity conservation and sustainable management of ecosystem services, such as carbon sequestration, may be aims at national or global levels; however, they may be out of the scope or underrepresented in local level decision-making if the benefits of conservation or ecosystem service management are mainly received at higher hierarchal levels. Whereas this interaction is very obvious in the case of government levels (e.g. local, regional and national governments as well as international agreements), it may also be the case for business. Subsidiaries from multinational companies are often controlled by headquarters situated at the other end of the world and decisions taken there may be subject to different incentives and optimisation criteria than those established at the local level of the subsidiary. Hence, rules established by business associations, stock exchange management and rating agencies may be very influential in the way biodiversity and ecosystem services are considered in business reporting and thus as a criteria for investment decisions by private enterprises.

3.3.5 Considering cultural and constitutional settings

Responses to biodiversity loss range from emotional to utilitarian, and so do existing and potential efforts towards biodiversity conservation and sustaining ecosystem service provision. Following the TEEB approach (TEEB, 2010a) one could distinguish between two types of cultural perceptions and constitutional settings in relation to the benefits and values inherent to biodiversity and ecosystem services. These types will influence the appropriateness of different policy responses, especially (monetary) valuation of ecosystem services and the implementation of economic instruments to capture such values.

In some instances, societies have already recognised the value of ecosystems, landscapes, species and other aspects of biodiversity, e.g. where spiritual or cultural values of nature are strong. In such circumstances there is some type of governance for securing these values in place, ranging from religious taboos or traditional sustainable management practices to a legally enforced network of protected areas. Irrespective of the actual mechanism applied these governance structures rest on a sense of collective heritage or patrimony, a perception of shared cultural and social value being placed on treasured landscapes, charismatic species or natural wonders. In such situations, government intervention and in particular the use of economic instruments is not a matter of undisputed consensus. Indeed it may even be counterproductive as it can be seen as contrary to cultural norms or failing to reflect the plurality of values associated with biodiversity and ecosystem services (TEEB, 2010a: 11).

However, a broad range of ecosystem services and features of biodiversity still lack the appropriate recognition of their inherent value and importance in policy and governance mechanisms. This is especially obvious in the negative externalities associated with the (unsustainable) utilisation of marketable ecosystem services, e.g. food production or extraction of raw materials that may have major impacts on biodiversity and the provision of other ecosystem services. Moreover, unsustainable practices may even be spurred by environmentally harmful subsidies. In such cases, government intervention or non-state regulation (e.g. via certification, see Kaechele et al., this report) is inevitable to correct for the market and / or policy failures. Whereas this does not imply the exclusion of certain instruments, economic instruments merit special consideration for capturing the so far unrecognised values of biodiversity and ecosystem services.

Moreover, constitutional settings may act as restrictions to the use of certain instruments. Exemplarily, in European agricultural policy the use of negative incentives is protected by the fiscal sovereignty of EU member states, whereas the use of positive incentives requires justification not only by WTO law, but also by EU state aid law. Hence, in many instances, the resulting policy mix consists of the EU nature conservation law which only incorporates considerations of economic efficiency in some cases, and some economic instruments in the Common Agricultural Policy which could be significantly improved with regards to their effectiveness in protecting biodiversity (see Klassert and Möckel, forthcoming).

3.3.6 Conclusion

As outlined above, there are major challenges for biodiversity conservation and sustaining ecosystem service provision. These range from the ecological characteristics of biodiversity and the vagueness of the ecosystem service concept to the variety of drivers of biodiversity loss and ecosystem degradation, the affected and impacting actors at different levels, scales and in different societal groups, to the cultural perceptions and constitutional settings of biodiversity and ecosystem service governance. Nevertheless, it has become obvious that biodiversity conservation will require a different policy approach than that of managing marketable provisioning services and what will be applicable to non-marketable regulating or cultural services. It is also clear that any policy employed has to deal with the great uncertainty and ignorance typical for the biodiversity-ecosystem service-human well-being interface.

Thus, the second step of our framework deals with choosing the instruments capable of dealing with the characteristics and specifics associated with biodiversity conservation and ecosystem service management and assessing existing policies for their comprehensiveness and performance regarding these challenges.

3.4 Second Step: Identifying gaps and choosing instruments for analysis

Whereas the first step of the proposed framework outlined the challenges and the context for a policy response to biodiversity loss and ecosystem service degradation, the second step is to identify gaps and to choose among the available instruments in the well-equipped toolbox. In this respect, it is necessary to firstly identify the policies already in place, as most aspects of biodiversity are already covered or at least influenced by existing policies (see 3.4.1). These policies will not always originate from environmental policies only, but might stem from different sectoral policies, e.g. agri- and silviculture, energy, transport or trade policy as well. Taking stock of existing policies may point to shortcomings, unaccounted trade-offs and blind spots of the currently applied instruments. Based on such assessment, policy makers may have two options or pathways to enhance the overall performance of the policy mix (see Figure 2 as well as Ring and Schröter-Schlaack, this report). On the one hand, they could aim at improving the existing mix of instruments by explicitly considering the effects of instrument interaction in fine grain design of single components of the mix (ex post analysis). On the other hand, policy makers may opt for introducing a new instrument to the existing mix in order to account for yet unconsidered aspects of the problem (ex ante analysis). This may include e.g. actors, activities, or sectors so far not explicitly addressed or the acknowledgement of recently evolved ecological knowledge.

Secondly, it is essential to assess the strengths and weaknesses of the individual instruments available as some of these policies may be better suited than others in addressing the challenges outlined above (see 3.4.2). And lastly, if instruments are applied simultaneously they will not only work towards the desired policy goal, e.g. biodiversity conservation, but they may also interact and thereby influence the performance of the policy mix (see 3.4.3). Thus, it is necessary to reconsider the classifications of instrument interactions available (see Ring and Schröter-Schlaack, this report), identify the functional role of each approach within a policy mix and choose complementary instruments to the policies already in place (see Figure 2).

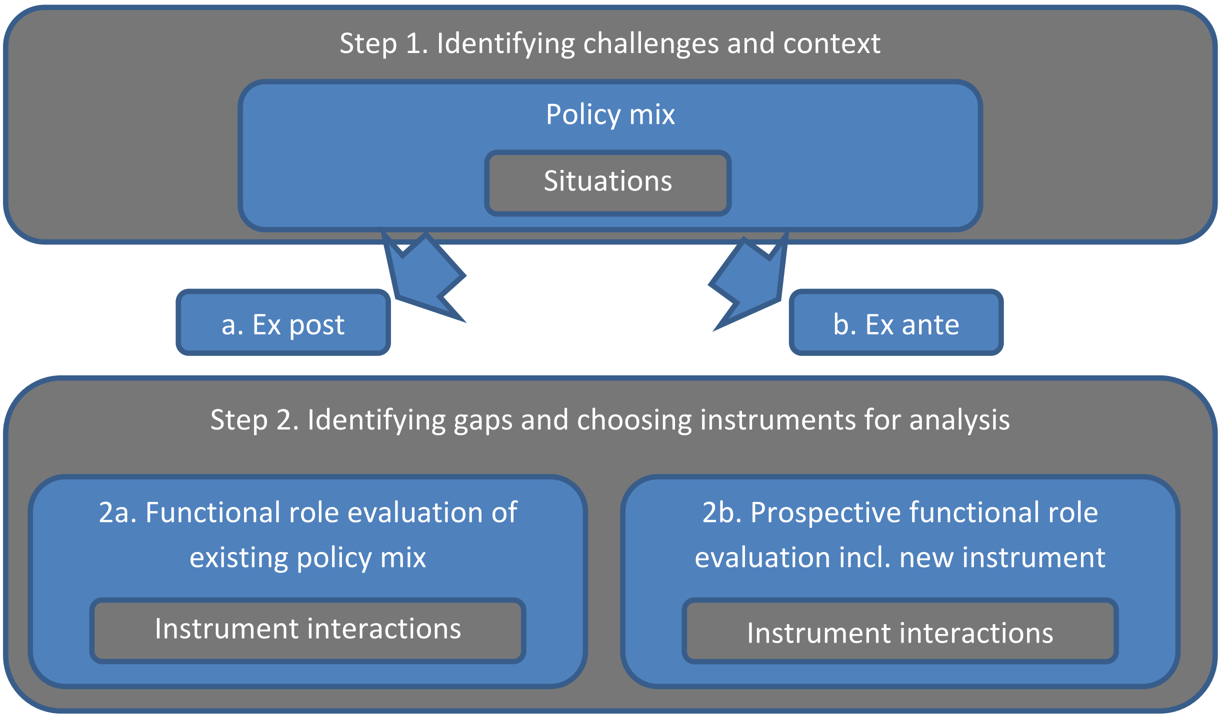

Figure 2. Distinguishing between ex post and ex ante analysis pathways

3.4.1 Assessing policies in place

In most instances, biodiversity and ecosystem services are already covered or at least somehow impacted by existing policy instruments. The potential bandwidth is shaped by the heterogeneity of landscapes and associated land uses and ranges from broadband interventions like land-use planning and environmental regulation for pollution to traditional nature protection measures, like protected areas or regulations for trade in endangered species and further on to specific regulations at regional and local levels, e.g. a PES-scheme within a catchment area.

Hence, it is advisable at the outset to assess existing policies against the challenges outlined in the first step of our framework and to analyse the types of interaction that might spur or hamper their performance (see Ring and Schröter-Schlaack, this report). In order to broadly assess the performance of existing policies one has to look at the five challenges identified above (see Box 2). Further guidance for policy assessment will be provided by the POLICYMIX project case study guidelines to be developed in work packages 3 to 6 (see also section 3.5 of this chapter).

In such analysis it is essential to keep in mind that policies influencing biodiversity conservation and ecosystem service management will not only originate from environmental policy. Ecosystem services, and thus conservation of critical biodiversity levels, are important inputs to a wide range of human activities and in turn object of many different sectoral policies as well. In most instances these sectoral policies will have negative impacts on biodiversity conservation. This may be either directly, e.g. through infrastructure development, clear cutting of forest to provide land for agriculture or the extraction of raw materials; or indirectly, e.g. through elevating the indirect drivers of loss, such as subsidies for energy or fuel that will consequently lead to higher impacts on the environment.

Box 2. Assessing existing policies against the challenges for biodiversity conservation and ecosystem service management

- Firstly, do the policies in place adequately address the irreversibility of biodiversity loss as well as thresholds of ecosystem resilience that – once crossed – will result in a failure of the ecosystem to deliver its services? For example, are there any instruments applied that are able to effectively restrict ecosystem degradation (e.g. protected areas, offsetting requirements for environmental impacts, or quantity-based instruments). Moreover, is the substantial uncertainty and ignorance in our knowledge on these ecological thresholds considered (e.g. are conservation and management goals adaptable to new knowledge or results from biodiversity monitoring and assessments)?
- Secondly, do the instruments in place address the trade-offs between biodiversity conservation and ecosystem service provision on the one hand and between different ecosystem services on the other? For example, are there requirements to systematically identify, consider or offset environmental impacts associated with development projects?
- Thirdly, are the drivers of biodiversity loss and ecosystem degradation identified and addressed by existing policies? Do existing instruments, e.g. subsidies, establish perverse incentives and amplify negative impacts?
- Fourthly, are all relevant actors addressed or who is missing? Are the relevant governance levels (local to global), domains (e.g. public to private) and the different modes of decision-making within these governance spaces addressed?
- And lastly, what is the scope of new instruments judged on available experience of policy-makers and policy-addressees and the overall attitude of the society regarding biodiversity conservation, ecosystem service management and public regulation? For example, is there any experience available at international, national or regional level on how to successfully introduce certain instruments that may provide guidance? Or on the opposite, are there any well-known failures in earlier implementation processes or in other policy sectors that may hamper the introduction of certain instruments for biodiversity conservation and / or ecosystem service management?

Furthermore, policies in place shape what further instruments can be added. On the one hand, (positive) experience with some type of instrument will facilitate its implementation in other fields of environmental policy. Thus, widening existing management standards to cover yet unconsidered management aspects or broadening permit trading to include actors from other sectors as well may sometimes be easier than implementing a new instrument. For example, permit trading is much more familiar to policy-making in the US than in Europe, where environmental taxes dominate as preferred economic instruments in environmental policy (see e.g. Hansjürgens, 2005). Hence it can be expected that introducing permit trading to European legislation will be more difficult than introducing environmental taxes. On the other hand, bold use of some instruments might hamper employing it further to achieve more ambitious (conservation and management) goals. For example, increasing 'direct regulation' of activities, e.g. tightening management standards or expanding protected area networks may be controversial, since these measures are deemed to be more prescriptive and costly for policy addressees than market-based approaches. Hence, relying only on 'direct regulation' may hamper the implementation of more ambitious conservation goals and management standards in political discourses. Lastly, already existing instruments are in some instances characterised by strong persistence. I.e. even in the light of new knowledge of their (unintended) effects they are not abolished or reformed as they might serve strong vested interests of certain actors or stakeholder groups, e.g. from other sectors. The long-lasting but still unsolved struggle about environmentally harmful subsidies is a distinctive example in this regard.

Against this background, we conclude that the assessment of the existing instruments is an integral part of designing policy mixes and deciding upon which instruments of the existing mix to reform or abolish and which instrument to add (see also section 3.5 and guidelines developed by work packages 3 to 6 of the POLICYMIX project).

3.4.2 Context-specific strengths and weaknesses of instruments

After assessing the existing instruments and identifying their shortcomings, potential ways to reform or supplement the existing policies have to be identified that enhance existing policies in addressing the challenges of biodiversity conservation and ecosystem service management outlined above. In a nutshell, what instrument to pick from the overall menu of instruments in order to enhance the performance of the policy response?

Every instrument works by a different mechanism and thus some may be better suited than others to address the challenges of biodiversity conservation and ecosystem service management. For instance, 'direct regulation' is deemed to be effective in securing a safe minimum standard of biodiversity conservation. In contrast, the main argument in favour of economic instruments is that they allow compliance costs borne by policy addressees to be reduced. Judged upon the characteristics identified in the literature reviews, this subsection strives to identify hierarchical relationships of instruments, i.e. are there instrument that will form the centre of the policy response and are thus leading instruments while others play a complementary role? As this judgement will be highly context-specific some of the main challenges for biodiversity conservation and managing ecosystem services are separately addressed below.

Dealing with uncertainty and ignorance

Notwithstanding the difficulties for policy design in the face of uncertainty and ignorance about the resilience of ecosystems, thresholds in ecosystem change and the biodiversity-ecosystem service nexus there are some general pathways towards instrument design. Firstly, it seems to be wise to err on the side of caution as unintended or unpredicted consequences of human activities may cause irreversible biodiversity loss and associated harm to human well-being (TEEB, 2010a: 26). Hence, the policy response to biodiversity loss should include instruments that are able to protect a 'safe minimum standard' of biodiversity conservation independent of dynamically evolving cost-benefit considerations of the addressed actors. 'Direct regulation' and the establishment of protected areas, no-take zones or prohibitions for the use of certain products and substances heavily impacting biodiversity are thus a key component of the policy mix (Hansjürgens et al., 2011b). However, it is still another challenge to define a 'safe minimum standard', though this is essentially an ethical judgement about the socially acceptable margin of safety in the exploitation of the natural environment (Perrings and Pearce, 1994) likely not to be captured in economic cost-benefit-considerations. And moreover, yet another challenge lies in the task of deriving priorities for the designation of protected areas, whether their size and spatial distribution should focus on preserving species, habitats, landscapes or ecosystem services provision (e.g. Brooks et al., 2006; Chan et al., 2006; Naidoo et al., 2006; Nelson et al., 2009; Rusch et al., 2011).

Secondly, uncertainty and ignorance may call for multiple instruments applied simultaneously. Whereas redundancy of instruments or their overlap in addressing actors, drivers or pressures of environmental threat are in general looked at with scepticism or straightforwardly rejected as causing inefficiencies (OECD, 2007: 27), it may indeed act as an insurance against knowledge gaps, policy or implementation failures in the case of biodiversity conservation (OECD, 1999: 12). When the drivers of biodiversity loss and ecosystem degradation are pervasive and cross-cutting through all sectoral policies (Barbier et al., 1994: 182) (see also below), the applied policies should themselves be pervasive and capable of filtering through the entire economic system (Gunningham and Young, 1997: 271). In this perspective, some overlap of instruments, i.e. instruments addressing the same actors but impacting on different biodiversity aspects and vice versa, and redundancy within the biodiversity policy portfolio, i.e. instruments addressing the same actor and its impact on the same biodiversity aspect, seem to be inevitable (and desirable even from an economic angle on instrument choice).

Lastly, as our knowledge on ecological functioning, resilience and critical thresholds will evolve over time, an important component of the policy portfolio for managing biodiversity will be motivational, educational and informative tools to facilitate attitude change and the alteration of traditional management practices (Stoneham et al., 2003).

Lacking property rights

As noted before, due to their public good characteristic, biodiversity is lacking clearly defined property rights, e.g. rights to use or access a resource or the right to compensation for income loss due to activities of other actors (see e.g. Hansjürgens et al., 2011a). Only for some ecosystem services, in particular marketable ones like food, timber and raw materials such property rights are defined and at – least in theory – also enforceable. Nevertheless, many aspects of biodiversity remain public domain. For example, rights of countries to participate on revenues from the sales of pharmaceuticals derived from bio-prospecting are continuously contested. Moreover, as people cannot be excluded from the benefits of biodiversity conservation provided by a single patch of land, landholders cannot capture people's willingness to pay for conservation provided.

This in turn has implications for the potential use of policy instruments, in particular economic instruments. While these approaches can be applied as broadband treatment to reduce direct and indirect drivers of biodiversity loss (e.g. taxes on emissions or the use of fertilizers), there appears to be little scope for the use of taxes or subsidies for providing conservation of biodiversity when property rights are not sufficiently defined (see inter alia Stoneham et al., 2003). Thus in these circumstances incentive measures will fail to provide effective conservation or management action, because the gains from conservation efforts cannot be captured by private individuals. 'Direct regulation' may bridge that gap by providing some basic property rights in differentiating between legally allowed and illegal activities, for example by stating whether a landholder is allowed to cut trees on his property or not. Thereby 'direct regulation' acts as reference point upon which market-based instruments can build, e.g. trading schemes (e.g. to exchange rights to cut), biodiversity offsets and banking (e.g. cut is only allowed if compensation is provided) or price-based instruments (e.g. cutting taxes or stumpage fees).

Spatial targeting of instruments and accounting for additionality

What is more, many economic instruments are in the first place unable to control for spatial allocation of compliance or conservation activities (see e.g. Nuissl and Schröter-Schlaack, 2009). Indeed, it will be necessary to either fine-tune the design of incentive measures, e.g. by spatial bonuses (e.g. Wätzold and Drechsler, 2005), or to couple them with 'direct regulation', e.g. zoning approaches to spatially target conservation efforts (see for coupling tradable permits and zoning inter alia Hansjürgens and Schröter-Schlaack, 2008; Henger and Bizer, 2010; Tietenberg, 1995). Lastly, many incentive-based approaches to foster private conservation efforts require a baseline of minimum management and conservation action to specify additional activities which are then eligible to remuneration. Very often, e.g. in Europe, such baseline is provided by 'direct regulation', i.e. legally defined management standards such as good practices in agriculture or forestry (Hansjürgens et al., 2011a).

What type of ecosystem service in question?

Due to the vagueness of the ecosystem service concept it is imperative to clearly distinguish what kind of ecosystem service is addressed and in which way this service is utilised. For marketable provisioning services economic instruments to correct market failures and to internalise external effects merit special consideration. On the contrary, for non-marketable regulating or cultural services economic instruments may be less appropriate, either due to their failure to safeguard a critical threshold to sustain ecosystem service provision or due to the ethical conviction that is it not appropriate to let the market regulate the provision of such services. Hence, whereas critical conservation thresholds to avert imminent dangers should be safeguarded by highly effective measures, such as 'direct regulation', more flexible market-based instruments merit consideration for controlling utilisation rates of ecosystem services beyond maintaining critical stocks. Moreover, it seems advisable to apply a number of instruments simultaneously, some directly targeting the ecosystem service in question, e.g. subsidies for environmentally food production and eco-labelling of organic food to inform and guide consumers in their buying decisions.

Summary

To conclude, it is obvious, that efforts to reduce drivers of biodiversity loss and ecosystem degradation cannot only be limited to biodiversity policies or environmental policies per se but have to be concerted by multiple initiatives across all sectoral policies (Hansjürgens et al., 2011b). The ultimate goal is to mainstream the yet unconsidered benefits of biodiversity conservation into these policies (TEEB, 2010a). 'Direct regulation' will have to play a crucial role in safeguarding a minimum level of biodiversity to avoid crossing critical thresholds of ecosystem functioning. Economic instruments merit particular consideration for managing marketable ecosystem services, and sustainably using ecosystem services within safe margins that do not endanger ecosystem functioning. Motivational, educational, informational instruments are always an important component of the policy mix as they raise awareness for biodiversity conservation and the consequences of continued loss of biodiversity and ecosystem service degradation, enhance acceptance of policies, and increase participation in voluntary conservation and management measures. Moreover, overlap of instruments constitutes an insurance against knowledge gaps, policy and implementation failures and should thus not be treated as generally inefficient over-regulation. The spatial heterogeneity of biodiversity conservation and ecosystem service provision potential often requires a mix of instruments to be applied. Economic instruments may link to regulation or planning (eligible areas, e.g., PA), or provide spatial bonuses in areas targeted for special conservation efforts. The performance of 'direct regulation' can be supported by economic instruments when actors are incentivised to provide conservation and management action beyond regulatory minimum requirements.

Following these insights, there is no universal hierarchy between 'direct regulation' and economic instruments. Even within the same class of ecosystem services, the provision of some services may be more threatened than others, thus requiring different characteristics of the policy response regarding effectiveness, duration till effect and efficiency. For ecosystems with an imminent threat of losing the ability to further provide their services "pure preservation measures will be required … These generally require restrictions on access to the biological resources or ecosystems, such as through the development of natural parks to protect ecosystems or habitats, or implementing regulations which prohibit the harvesting or other use of a particularly threatened or endangered species" (OECD, 1999: 11). Hence, economic activity should be bound to levels that do not threaten the resilience of the system (Perrings and Opschoor, 1994: 10). On the contrary, wherever "biological resources can be used for economically productive purposes in a sustainable manner…the most appropriate incentive measures to ensure that their use does not lead to biodiversity depletion are the creation of markets and the assignment of well-defined property rights to realise the full private benefits of the resources, in combination with regulations and standards to prescribe the allowable levels and types of use" (OECD, 1999: 11). Thus, within the boundaries of ecosystem resilience markets should be permitted to work (Perrings and

Opschoor, 1994: 11) and public policies should focus on facilitating market creation (e.g. by setting up tradable permit markets) or on correcting existing market failures by internalising external effects (e.g. by the design of liability rules, offset mechanisms, taxes, or fees to incorporate environmental costs into private decision-making).

3.4.3 Recognising interaction of instruments

The performance of the existing instruments may also be spurred or hampered by the interactions between the instruments. As Ring and Schröter-Schlaack (this report) have outlined, there are some instances where instrument interaction and overlap can be judged as productive, whereas there are other situations where instrument interplay and overlap may be counterproductive. Thus, after analysing the existing mix and the identification of the gaps and shortcomings of current policies on the one hand, and becoming clear about context-specific strengths and weaknesses of the alternative policy approaches on the other hand, now the interaction of different instruments gets in the focus. There are several classifications on instrument interaction within the literature. Exemplarily, Gunningham and Sinclair (1999) distinguish between four types of relationships between instruments: inherently complementary (positive) interaction, counterproductive (negative) interaction, sequential relation, and context-specific combinations. These will be looked at in more detail in the following.

Complementary, positive interaction

Inherently complementary combinations will enhance the effectiveness and efficiency of the employed instruments. They do so by addressing different aspects of the problem, e.g. different drivers of biodiversity loss and ecosystem degradation, different actors, e.g. private vs. public actors or increase compliance of actors by reducing moral hazard and rectifying information asymmetries between parties. Gunningham et al. (1998: 427 ff.) present a range of examples for such positive instrument interaction, inter alia:

- Informative instruments to rectify or compensate for information asymmetries e.g. between regulator and policy addressee are deemed to improve performance of the policy mix.
- Voluntary agreements will complement 'direct regulation', in particular where levels of environmental performance 'beyond compliance' are desired and (at least some) policy addressees are willing to voluntarily restrict their environmental impacts. All other policy addressees still have to comply with the baseline provided by 'direct regulation'.
- Performance-based management standards imposed by 'direct regulation' will be assisted by supply side incentives, e.g. tax concessions or soft loans for preferred technologies.
- 'Direct regulation' may even positively interact with broad based economic instruments, if they target different aspects of a common problem. For example producers may be forced to comply with a certain production standard, whereas consumer will be stimulated to buy biodiversity-friendly products if the price of conventionally produced goods is additionally surcharged by a tax.

Counterproductive, negative interaction

There are also inherently counterproductive interactions that diminish efficiency and effectiveness of the employed instruments. They do so by reducing the flexibility of policy addressees in choosing abatement options (e.g. across different technologies or across time) or by duplication of the same mechanism of action. Gunninghem et al. (1998: 437 ff.) also present a range of examples for these negative interactions, inter alia:

- 'Direct regulation' limits the choice of policy addressees in making individual decisions on the way to reduce environmental impacts with lowest costs, which is the key prerequisite for incentive-based approaches to realise least-cost-solutions. Hence the regulatory outcome of combining both approaches may be sub-optimal as the marginal abatement costs between policy addressees are not fully equalised.

- Similarly, technology-based standards (e.g. certain cutting techniques in forestry management) may reduce the flexibility of policy addressees to meet benchmarks set by performance-based standards (e.g. safeguarding habitats for endangered species) and thus reduce the efficiency of the latter.
- When incentive-based instruments, such as stumpage fees, habitat banking or offsets are applied at the same time as liability rules, then the same activity is subject to two different price signals (fee, permit price, offset costs vs. tort claims). This will be redundant at best or counterproductive at worst.

Sequencing or path dependency of instruments

In some instances, sequencing of instruments will have positive impacts on the different policy approaches. Such positive effects are typically realised by embedding the threat of introducing a more prescriptive or costly instrument later if performance of or compliance with the earlier introduced instrument is low. Again, Gunningham et al. (1998: 444 ff.) list some examples, namely:

- The credibility of self-regulation can be bolstered with a backdrop of 'direct regulation' – if policy addressees fail to deliver the promised improvements then authorities could step in to impose mandatory regulation.
- Similarly, the effectiveness of self-regulation may be facilitated by the threat of imposing a broad based economic instrument, e.g. taxes on pollution or permit trading, if there are no improvements. As these additional measures are then implemented across all affected policy addressees there is also an element of certainty and credibility of self-regulation within the group of policy addressees.

There may be also a sequencing of instruments in temporal terms when an existing instrument is reformed to consider new information about the ecological status of an ecosystem (e.g. a wetland, watershed etc.) or the consequences of traditional management approaches (e.g. cutting practices in forestry or mowing regimes in agriculture). For example, Costa Rica's PES-schemes evolved gradually from a forestry subsidy aimed at offsetting costs involved in establishing and managing forest plantations introduced in the late 1970s to the proactive payment schemes of today (Daniels et al., 2010; Watson et al., 1998). Similarly, the agri-environmental schemes in the EU are a subset of the more traditional agricultural subsidies-system. The share of these conservation and management action-oriented payments on total EU spending on agriculture subsidies has considerably grown over the last decade (European Parliament, 2008). In both cases, policy makers made use of an existing instrument but realigned their targets to more recent priorities.

Context-specific combinations of instruments

Lastly, Gunningham et al. (1998: 446) distinguish instrument combinations where the outcome will depend on the particular context in which the two instruments are applied. For example, informative measures (like forest certification) and 'direct regulation' (like forestry management standards) are deemed to act complementary as they are addressing different actors, e.g. the latter is targeted at producers whereas the former is designed towards consumer. However, as Kaechele et al. (this report) state, the presence of multiple norms (certification rules vs. regulatory management standards) within a given country or region may lead to confusion on part of the consumers, and may thus undermine incentives to comply with the more rigorous standards (especially when enforcement of legal standards is weak).

These context-depending instrument combinations give rise to establish a 'policyscape' concept. Depending on the particular context of a landscape, e.g. characteristics of relevant aspects of biodiversity, different land uses in a land-use mosaic direction of benefit flow from ecosystem services as well as the institutional-political setting, such as the mix of actors, affected administrative structures and already existing policies and instruments, a specific ‘policyscape’ may evolve (Barton et al., 2011). While such ‘policyscape’ will basically follow the more general conclusions regarding the comparative advantages of certain instruments, the context specificities of each individual case will shape its ultimate composition. Thus, building on the classification proposed by Gunningham and Sinclair (1999) the POLICYMIX case studies will be particularly focussing upon analysing and describing the respective ‘policyscape’, thereby providing positive insights to the normative classification outlined here.

3.4.4 Conclusion: Assessing the functional role of instruments in policy mixes

To conclude, there are three main determinants that influence the composition of the mix and that define the functional role of different instruments within the policy mix, namely the performance (and composition) of the existing policy (mix), the context-specific strengths and weaknesses of the individual instruments and lastly the interaction of the instruments within the policy mix.

Firstly, an analysis of the performance of existing policies will point to their shortcomings regarding the challenges of biodiversity conservation and ecosystem service management. Moreover, experience with existing policies in place shape what further instruments can be added more easily.

Secondly, the different strengths and weaknesses of instruments are of different importance for different conservation and management goals. For instance, ‘direct regulation’ is deemed to be effective in securing a safe minimum standard of biodiversity conservation and critical ecosystem service provision. In contrast, the main argument in favour of economic instruments is that they allow compliance costs borne by policy addressees to be reduced, e.g. in sustaining provision of marketable ecosystem services.

The third determinant for the role of individual instruments is how the additional instrument will interact with existing policies. Each instrument works by a different mechanism, either prescribing certain actions (‘direct regulation’), incentivising positive actions (PES, subsidies, tax reliefs, fiscal transfers), penalising negative impacts (offsets, taxes, permit trading) or providing information to stipulate motivation and self-regulation (certification). Some of these mechanisms are deemed to be complementary, e.g. facilitating policy instruments by informative measures. Others are deemed to be counterproductive, e.g. limiting the compliance options of policy addressees by ‘direct regulation’ may restrict the flexibility inherent to economic instruments and will thus limit the potential cost savings from applying economic instruments. Ultimately, a ‘policyscape’ depending on the specific characteristic of the landscape, its uses and the associated formal and informal institutions can be drawn.

Hence, depending on these three aspects, instruments will have different roles to play within a policy mix. They may either be the leading approach, often introduced as initial regulatory impulse and amended by other policies to avoid negative side effects. On the contrary, an instrument may also be applied later to facilitate already existing policies. For example, ‘direct regulation’ is very often the pioneering approach to reduce environmental loads and to safeguard biodiversity conservation. It may be augmented later by economic instruments to reduce opportunity costs of implementing more ambitious conservation goals or by informative measures to enhance compliance and reduce costs for monitoring and enforcement. Nevertheless, there may be also situations, where economic instruments are the main policy in place, e.g. taxes to correct for externalities in utilising marketable ecosystem services. Later on, the economic instrument may be augmented by ‘direct regulation’, e.g. zoning, to spatially allocate compliance activities to biodiversity hot spots.

3.5 Third Step: Policy evaluation and design

The last step of the proposed framework now turns the focus to the evaluation and design of single instruments, i.e. how to improve an existing or design a new instrument so that the additional value of the relevant instrument to the existing policies is maximised? Although there is ample (economic) literature on instrument choice and design, these contributions very often strive towards developing optimal single instrument policies. However, as outlined above, the characteristics of and challenges associated with biodiversity conservation and ecosystem service management will in many instances require the simultaneous use of multiple instruments. And whenever more than one instrument is implemented, the interaction of instruments is of fundamental importance for overall performance of the policy mix. Against this background, the overall aim of instrument evaluation and design is shifted towards the specific role of single measures within a policy mix and how single instruments facilitate the performance of the overall policy mix.

Depending on whether policy makers choose to reform an existing instrument or to introduce a new one, two different pathways for single instrument design can be distinguished in step 3 (following Ring and Schröter-Schlaack, this report: 15; see also Figure 3 below):

a. Ex post analysis: to improve the success of the existing policy mix, impact evaluation of one selected policy instrument against the background of the other instruments in the mix is performed using criteria for single instrument analysis as well as using criteria for designing policy mixes;

b. Ex ante analysis: a new policy instrument is introduced against the background of already existing instruments and both the new and the existing ones form the policy mix. In this case, scenario evaluation may be used to design the new instrument regarding its performance as a single instrument, but also in terms of its additional value or conflict potential for the overall policy mix.

To develop policy recommendations we refer to the traditional evaluation criteria while moving beyond the core criteria of effectiveness and efficiency in economic analyses, and group them into four basic assessment categories: conservation effectiveness, cost-effectiveness, social impacts and policy legitimacy, and institutional aspects. When dealing with policy mixes, the ultimate goal for instrument design is no longer to develop first-best single policy solutions, but to optimise design regarding the functional role of the instrument in the policy mix. Building on the four categories of criteria mentioned and the classification of instrument interaction outlined in section 3.4.3, we distinguish four pathways on positive interaction between policies: a) to increase effectiveness or b) to increase efficiency of the policy mix by accounting for relevant benefits and costs, c) to enhance an equal and socially fair distribution of policy consequences, and finally d) to reduce transaction costs and institutional barriers associated with single instruments or the existing policy mix. All of these aspects are highly context-specific and so are the methods from various scientific disciplines needed to derive some concrete recommendations. Hence, the following paragraphs only shortly sketch the major challenges that need to be addressed. The subsequent work packages 3 to 6 of the POLICYMIX project will develop detailed assessment criteria for policy and larger governance analysis and recommendations regarding these aspects, thereby encompassing knowledge and techniques from natural science disciplines, such as biology and landscape ecology, to social sciences, such as economics, sociology and law.

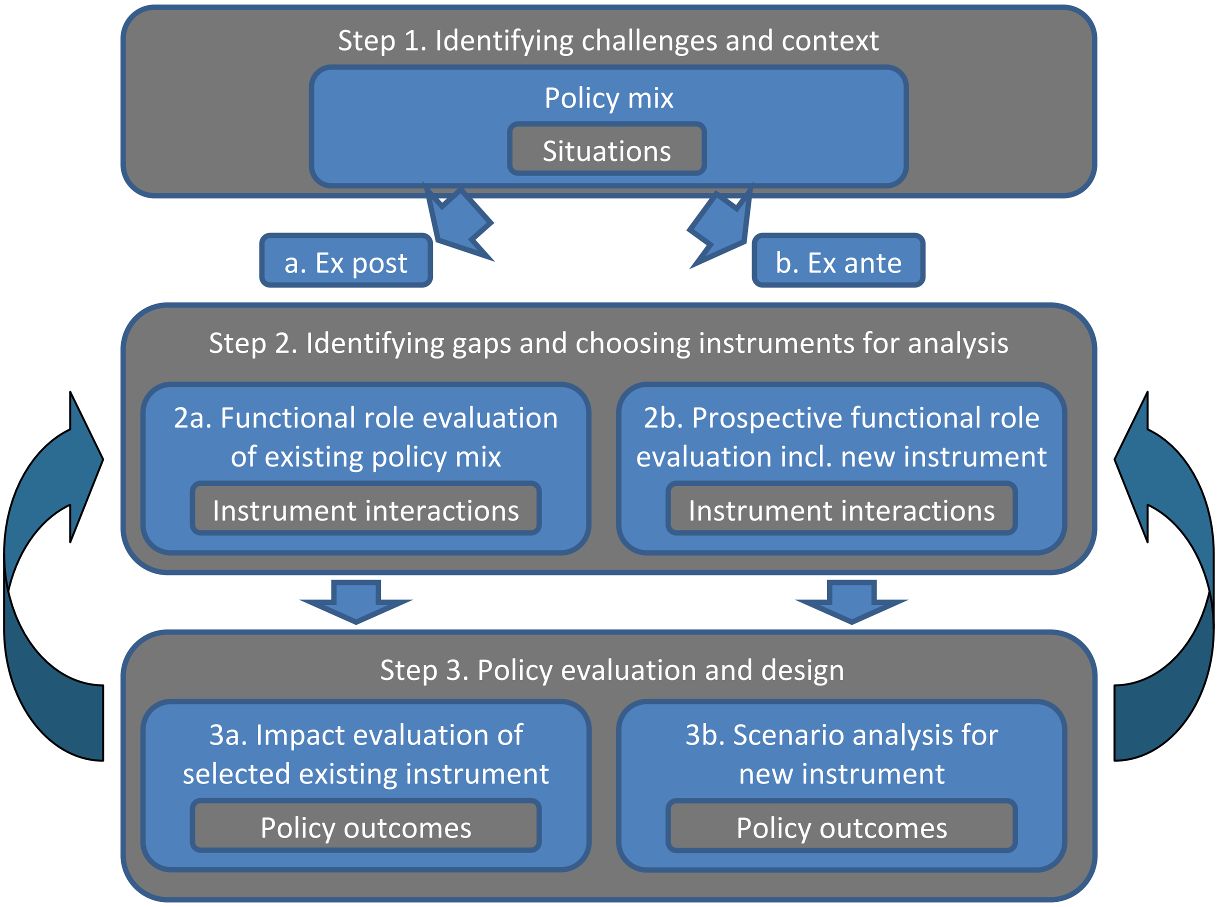

Figure 3. Policy mix analysis framework and pathways

Firstly, an essential focus is on the environmental effectiveness of the policy mix, i.e. how to design different instruments to assure that an environmental goal, e.g. specific biodiversity conservation targets, will be achieved effectively, timely, and enduring? To answer such questions and to derive recommendations for the evaluation and design of policy instruments a methodological framework to quantify conservation gains and losses produced by the various conservation instruments is needed. Thus, work package 3 on 'Ecological effectiveness of policy instruments' will review methods and models for quantifying gains in biodiversity conservation and in ecosystem service provisioning based on biodiversity inventories as well as land-use and land-cover data. Depending on available data, a tiered approach of indicators for biodiversity conservation gains and ecosystem service provisioning at different spatial scales will be developed (see Rusch et al., 2011).

Secondly, another important angle of analysis aims at potential cost savings of policy mixes enabled by improving existing or adding further instruments, i.e. how to improve the policy mix to lower the cost of goal attainment? In order to derive such recommendations one needs to achieve clarity about the spatial and temporal distribution of benefits and costs of conservation and the transaction costs associated with the different instrument alternatives. Work package 4 on 'Economic benefits and costs of economic instruments and their implementation' develops guidelines for biodiversity valuation and the assessment of the economic benefits and costs of policy instruments. Moreover, a cost-accounting framework is developed to improve transparency and consistency in economic policy instrument implementation and better integrate the associated transaction costs into decision-making (Brouwer et al., 2011).

Thirdly, and closely connected to the question of efficient policies are the issues of distributive and procedural justice. Although the social distribution of costs and benefits is typically outside traditional cost-benefit-analysis, when focusing on a pure economic angle in relation to instrument choice (see for the Kaldor-Hicks criterion as one of the underlying rationales of cost-benefit-analysis: Hicks, 1939; Kaldor, 1939), it is of major importance for acceptability and implementation of instruments in practice. Moreover, if policy addressees conceive policy outcomes as unfair or have the impression their claims are ignored in policy implementation this might have severe effects on policy success in the form of. for example, obstacles for political acceptability of instruments, court complaints and high transaction costs of policy enforcement. In the end, not considering processes (procedural justice), impacts on well-being (distributive justice) and legitimacy (sense of justice) early on in the analysis and design of policy instruments (Grieg-Gran et al., 2011) may result in lower effectiveness of the policy (due to low compliance rate or high incentives for illegal behaviour) and high societal costs (due to high transaction costs of implementing policies and enforcing the rules). Hence, work package 5 on 'Social impacts of instruments and enhancing policy legitimacy' reviews the application of social impact analysis in policy evaluation and instrument design, including both distributional (outcome) and fairness-related (process) issues, in order to provide relevant assessment criteria for policy analysis in the POLICYMIX case studies. Based on experiences, best-practice guidelines for social impact and legitimacy analysis of policy instruments will be specified.

Fourthly, all policy instruments are embedded in a number of existing formal as well as informal institutions. As these institutions frame and shape the way policy instruments are introduced, designed, and implemented, they are important contextual factors. Hence, relevant institutions need to be considered in evaluating existing and designing new instruments in order to facilitate acceptability and implementation of reformed or additional instruments. Spoken in economic terms, the aim is to reduce transaction costs and other institutional barriers associated with certain instruments or the existing policy mix. For the purpose of the institutional analysis to be conducted, all existing policy instruments except those instruments that are the target of assessment belong to the institutional context, i.e. they can be considered institutions, and should be analysed if they are relevant for the assessment (Similä et al. 2011). The description of institutions should preferably be systematic enough to allow some level of comparison, and ideally, should encompass empirical evidence of the presence/absence and type of institutions as well as their influence in policy design, implementation and impact. Work package 6 of the POLICYMIX project on 'Institutional and legal options and constraints' provides more detailed guidelines and recommendations for analysing the importance of institutions in evaluating and designing instruments and thus policy mixes (Primmer et al., 2011).

Finally, depending on the policy-relevant outcomes of the evaluation and design of instruments in step 3, it may be necessary to reconsider the functional role of the relevant instruments in the policy mix (step 2) (see Figure 3).

4 Conclusions

Real-world policies and environmental policy in particular are characterised by the existence of policy mixes. This holds especially true for policy responses to the ongoing biodiversity loss and the associated degradation of ecosystems' ability to provide ecosystem services. Despite this observation, most of the literature on instrument choice has focused on the analysis of individual instruments rather than policy mixes. Building on the existing literature on policy mixes and a number of reviews on selected individual policy instruments, this chapter has developed a generic framework for assessing instruments in policy mixes for biodiversity conservation and ecosystem service provision (see Figure 3).

For assessing instruments in policy mixes and deriving recommendations for the composition of the mix and the design of individual instruments in these mixes we propose a tiered three-step approach. The first step is to analyse the policy problem to be addressed, i.e. in our case biodiversity conservation and forest ecosystem service management. We have argued that there are at least five aspects that will be influential on the performance of policies in this regard: The policy response must a) address the specific characteristics of the underlying problem, i.e. biodiversity conservation and ecosystem service management and should b) cover all major policy objectives in this field. Moreover, it is necessary to c) comprehensively embrace relevant drivers and pressures on biodiversity loss and ecosystem degradation by d) actors at relevant levels and scales. Lastly, composition and performance will be e) dependant on the institutional framework, that will act both as a constraint and enabling condition for certain instruments.

Building on the identification of the challenges and context of the policy problem, the second step is to identify gaps and choose instruments of the well-equipped toolbox for biodiversity conservation and ecosystem service management. Here, it is necessary to become clear about the relationship, interplay and functional role of single instruments within a policy mix. In order to do so, one has firstly to identify existing policies and assess their performance against the challenges outlined in the first step. Building on such an assessment of the existing mix, policy makers have two options to deal with potential deficiencies. They may either choose to reform existing instruments or to introduce new instruments in order to account for yet unconsidered aspects that hamper full implementation or effectiveness of the existing policies. By identifying the working mechanisms of individual instruments (see Ring and Schröter-Schlaack, this report: Table 1), their strengths and weaknesses as well as the institutional prerequisites for their successful implementation one can reveal four basic types of interaction. Besides inherently complementary and counterproductive interaction, a sequential order of different approaches and lastly context-specific combinations of instruments can be distinguished. The main idea behind designing a policy mix (or a 'policyscape') would thus be to combine instruments that are deemed to interact positively (at least context specific) and are able to address the blind spots of the existing policy mix regarding the challenges outlined in step one. Moreover, in some instances it may be recommendable to remove instruments that have negative or unwanted side-effects, such as environmentally harmful subsidies.

The third and final step of the proposed approach is to evaluate and design individual instruments in such a way, that the additional benefits of improving an existing instrument or adding a new instrument to the existing mix is maximised. We suggest that this analysis can build on the same criteria as the evaluation of single instruments. However, the ultimate goal for instrument design is no longer to develop first-best single policy solutions, but to optimise design regarding the functional role of the instrument in the policy mix. In this regard, we distinguish four pathways on positive interaction between policies: a) to increase effectiveness or b) to increase efficiency of the policy mix by accounting for relevant benefits and costs, c) to enhance an equal and socially fair distribution of policy consequences, and finally d) to reduce transaction costs and institutional barriers associated with single instruments or the existing policy mix. The subsequent work packages 3 to 6 of the POLICYMIX project develop detailed guidelines in order to assess policy mixes and single instruments against these criteria and to derive recommendations regarding their design.

As in any other policy field, there will be no 'blueprint' for optimally designing a policy mix for biodiversity conservation and ecosystem service management as each country is different and relies on biodiversity and ecosystem services to a different extent (TEEB, 2010a: 31). Moreover, ecosystems may be in different stages of degradation and thus in different proximity to tipping points of critical ecosystem service provision. Finally, each country deals with a different set of policies already in place. Nevertheless, two recommendations on mainstreaming biodiversity conservation and ecosystem service management may apply in almost all cases, irrespective of the specific setting (TEEB, 2009: 31):

- The policy mix should not be limited to 'environmental' or 'conservation' policies but should also encompass other sectoral policies, like agriculture, energy or transport.
- A policy mix can be developed using a step-wise approach that starts with the more easily available opportunities.

By building upon the literature available this chapter developed a three-step iterative POLICYMIX framework for the assessment and design of instruments in policy mixes to explicitly consider the functional role of instruments in policy mixes when deciding about policy instruments for biodiversity conservation and ecosystem service management. It distinguishes two major pathways of analysis: a) ex post analysis to improve selected existing instruments in a policy mix regarding their functional role and employing impact evaluation for a detailed assessment of relevant design characteristics; b) ex ante analysis to identify relevant new instruments for a policy mix that address important challenges or functional roles thus far neglected, and employing scenario analysis for a detailed assessment of relevant design criteria. At the same time, our suggested framework offers linkages for the integration of ecological, economic, sociological as well as legal expertise as covered in the guideline development work packages 3 to 6 and required in the POLICYMIX case studies to be conducted in work package 7.

Work package 7 on 'Assessment of existing and proposed policy instruments for biodiversity conservation: case studies' will then implement the POLICYMIX framework in seven case studies, assessing the functional roles of instruments in policy mixes at national and local governmental levels. Work package 8 on 'Multi-scale comparative case study analysis and transferability assessment of economic instruments' ensures consistent comparison of major instruments and the methodological approaches and components developed in work packages 3 to 6 in comparative cross-case analysis at national and local level. Last, but not least, work package 9 on 'Methodological synthesis and policy recommendations' will revise guidelines for multi-scale policy mix assessment which were initiated in work package 2, detailed in work packages 3 to 6, and tested in the case studies. Policy mix design and case study transferability will be synthesised together with best-practice recommendations.

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