

Chapter 2

Managing Tropical Forest Ecosystem Services: An Overview of Options

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2.1 Introduction

Decision-makers have three types of choices to make regarding the management of natural resources and related ecosystem services (ES): (1) which ES to manage, (2) the quantitative and qualitative objectives associated with each ES (3) and the instruments for achieving these objectives. This chapter focuses on the final choice faced by decision-makers, that of selecting the proper instrument¹ from an array of options that includes standard environmental policies (e.g. land use regulations, taxes and subsidies) that are generally implemented by the State, as well as a broader set of alternatives (e.g. reallocation of property rights and joint management of common property resources) that can be implemented by stakeholders with or without involvement of the State. Choosing among alternative management instruments is difficult, first because information regarding their effectiveness and implementation costs is often missing or incomplete and, second, because managing ES often involves trade-offs with other policy objectives, such as economic growth, poverty alleviation or social equity (Cole and Grossman 2002; DeFries and Rosenzweig 2010; Lee and Barrett 2001).

¹ Throughout this chapter, we use the singular (instrument) with the understanding that sets of policy instruments (plural) may have to be deployed to achieve desired objectives; the effects and costs of these policy sets must be considered jointly.

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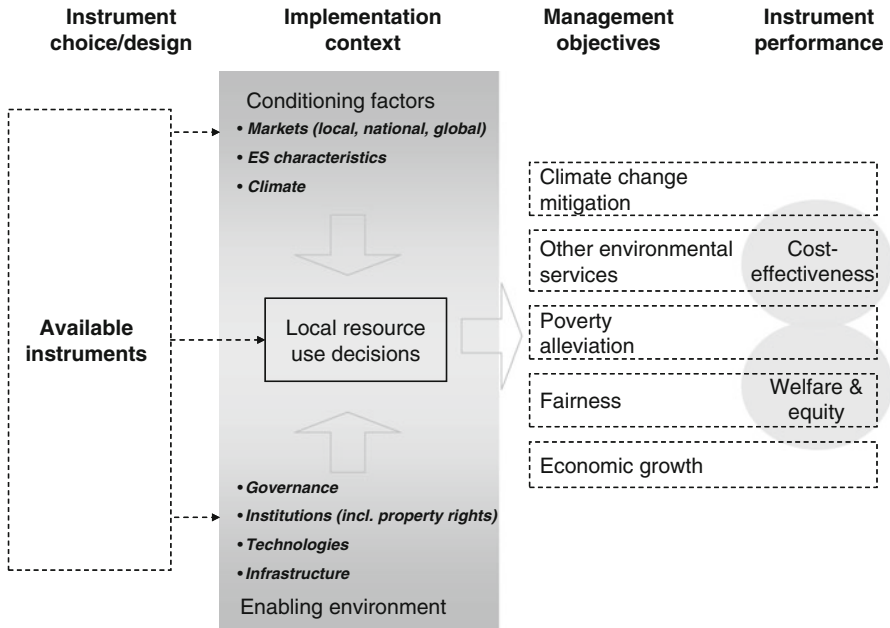


Fig. 2.1 A stylised impact pathway for ES management

The rationale for public policy attention (and perhaps action) in the context of ES with public good character is clear; environmental externalities, missing markets, information asymmetries, etc., suggest that without public sector interventions ES will be undervalued, overused and suffer from suboptimal levels of investment in many cases (Belli et al. 2001). Even ES that provide private benefits are sometimes underused or overused, for example, due to ‘conservation investment poverty’ (Vosti and Reardon 1997); these cases also merit public policy attention. However, since management is *never* costless, the existence of a market failure in the provision of ES is not sufficient to justify policy action. Decision-makers must understand the value or cost to ES beneficiaries of the effects of such a market failure and assess the effectiveness and costs of alternative options for addressing the problem. In the case of water pollution, for example, stakeholders may decide to invest in water treatment plants that substitute for natural water-purifying ecosystem services, especially if expanding natural purification systems displaces income and employment generating activities in upstream areas.

Therefore, taking any action at all requires that at least one instrument exists for which the benefits to society outweigh costs of implementing it over a defined time period; if this is not the case, then the socially optimal response is to take no action unless something changes this basic relationship. When several alternative instruments exist that pass this first fundamental test, attention is then focused on the relative cost-effectiveness of alternative management instruments. How a given policy instrument performs vis-à-vis alternatives depends crucially on the implementation context and design. To illustrate this, consider the conceptual framework in Fig. 2.1.

ES management fundamentally seeks to change natural resource use decisions in favour of a specific or collection of ES, for example, maintaining the climate-regulating function of tropical forests (i.e. reducing emissions from deforestation and degradation – REDD) to mitigate global climate change. Often environmental policies also have additional objectives, such as poverty alleviation and sustainable economic growth. Basically, three entry points exist to affect natural resource use decisions (Fig. 2.1). ES management can (1) change the rules of the game by affecting the conditioning factors of natural resource use, for example, market prices through certification of commodities produced with reduced impact on forest biomass (Veríssimo et al. 2005). Alternatively, decision-makers may (2) choose to influence natural resource use decisions directly, for example, through payments for ecosystem services (PES) (Ferraro and Kiss 2002), or (3) improve the enabling environment for sustainable local resource management, for example, through land tenure reform (Pacheco 2009).

Since local resource use decisions are influenced by conditioning (e.g. climate, natural resources and ES characteristics and markets) and enabling (e.g. institutions, infrastructure and technology) factors, the outcomes and performance (cost-effectiveness) of a given ES management approach also depend on these factors. For example, devolution of land rights to smallholders intended to improve community-based forest management may not be effective in reducing deforestation, where governance is weak and property rights are poorly enforced. Likewise, PES may exacerbate preexisting inequalities if land and pressure on forests is concentrated among only a few large landholders (Börner et al. 2010).

This chapter reviews the literature on ES management options in forested areas with two goals in mind.² Our primary objective is to provide an overview of instruments that have been used or proposed for managing tropical forest ecosystems and to assess their likely performance. Though we draw mostly on examples from the Amazon and Andes regions, many of the observations and conclusions are valid beyond this regional context. A second objective is to identify research needs. This chapter is organised as follows. The next section sets out a typology of management instruments. Section 2.3 identifies factors that influence the effectiveness of these instruments. Section 2.4 provides an assessment of the expected performance of specific ES management instruments. Section 2.5 concludes this chapter by providing implications for research, capacity strengthening and policy.

² We systematically screened over 600 peer-reviewed journal articles, research reports and institutional publications that dealt with the options for and the effects of environmental management. For each policy instrument category, key studies were analysed in more detail. Most publications deal with carbon, plant biodiversity and water-related ES; there were fewer studies of forest products, soil degradation and air pollution; few publications address specific and well-defined ES. We attribute this to the fact that the ES concept has only recently been widely adopted in the scientific literature, and that, with the exception of water, few ES-specific policy instruments are available. The Millennium Ecosystem Assessment (MEA 2005) provides one of the first, broad frameworks for defining and managing ES. A complete list of the reviewed literature can be obtained from the authors.

2.2 An Intuitive Typology of Management Instruments

Instruments to manage natural resource use and, hence, ES flows have been classified in many different ways. For example, Bayon (2001) distinguishes public-good-specific, incentive-changing and business options. Sterner (2003) divides the management toolbox into options to use markets, create markets, regulate use and engage the public. The MEA (2005) establishes categories of response options, such as legal, economic and social responses, among others.

Despite differences in the details associated with individual instruments examined in the literature, it is a common feature of all ES management instruments that they seek to influence human behaviour. Ideally, a decision-maker's goal is to 'adjust' human behaviour such that natural resources (and related ES) are used (or conserved) in socially optimal ways (Baumol and Oates 1988). In practice, decision-makers seldom know (or agree on) what this social optimum is and, even if they did, may not know how to achieve it. However, while we may not know what 'optimal' is, stakeholders in society do have strong preferences regarding natural resource management and ES, and we know that market forces (alone) will not deliver what most stakeholders prefer. Informed intervention thus requires an intuitive framework for choosing among multiple potential intervention options. If we classify management instruments according to how they attempt to influence human behaviour,³ three basic (and admittedly not strictly separable) mechanisms can be distinguished:

1. Establishment of general conditions that *enable behaviour* driven by private incentives to contribute to achieving a given ES objective (Enabling)
2. Provision of (specific) *incentives that change behaviour* in ways that contribute to achieving a given ES objective (Incentives)
3. Provision of (specific) *disincentives that change behaviour* in ways that contribute to achieving a given ES objective (Disincentives)

2.2.1 Enabling Measures

Management options in the 'Enabling' category contribute to establishing conditions that lead to the management of ES in more socially desirable ways without changing underlying incentives to resource users. In a sense, these 'enabling' options allow for ES outcomes that would emerge *if* economic agents' behaviours were not constrained by an unfavourable conditioning or enabling environment. Common constraints in developing countries include the lack of basic public services, such as health and education, or the enforcement of property rights. In addition, the private sector also often fails to deliver agricultural technologies with large public good components or

³ We emphasise the word 'attempt' because the intensity and duration with which a given instrument is used will, in part, determine its effect on human behaviour – for example, small price subsidies and short-term punishments may do little to change behaviour in the long term.

provide environmental education – in the former case, the private benefits of technological change that accrue to farmers are expensive to ‘collect’, while in the latter case, the benefits may not accrue at the individual level, so farmers will not be willing to pay for this service.⁴ Societal demand for improved ES may in many such cases represent an argument in favour of measures that ‘enable’ farmers to increase natural resource use efficiency, and thereby ES provision levels, through the adoption and correct use of improved production technologies.

However, implementing enabling measures will seldom guarantee a more socially acceptable outcome; no single enabling measure is likely to ‘remove’ all of the constraints that preclude the desired human behaviour (e.g. credit provision may not be sufficient to overcome the effects of insecure land tenure). Or in a less-constrained situation, an individual may choose behavioural options that do not involve the targeted ES (e.g. credit provided for soil-enhancing investments may instead be used to pay for educational services for children). Research has, nonetheless, shown that in some situations enabling measures have contributed to more efficient and socially optimal ES use or to less damaging ES modifications (Kuyvenhoven 2004). Enabling policy is often viewed as complementary measure needed for effective implementation of some incentive- and disincentive-based interventions (Auty and Kiiski 2002; Börner et al. 2010). Lastly, because enabling measures, by definition and design, aim to increase options available to resource users, they can have negative spillover effects on natural resources and related ES, for example, providing rural credit in forested areas can increase deforestation unless forests are not protected effectively by additional measures (Angelsen and Kaimowitz 2001; Vosti et al. 2002).

Studies all over the world have shown that situations of poorly defined or incompletely enforced property rights (i.e. resource tenure insecurity) and nonexistence of property rights (i.e. open-access situations such as unguarded or unmanaged common pool resources) motivate natural resource ‘mining’ strategies, that is, the rapid exploitation of ES and ecosystem goods in the face of uncertain opportunities for future use (Hotte 2001; Schuck et al. 2002). A frequently cited example of an enabling ES management option to deal with this problem is the transfer of property rights to natural resources or related ES from (say) federal control to lower-level administrative units or even local resource users (Agrawal and Gupta 2005; Persha et al. 2011). The effectiveness of this approach naturally depends on whether the resulting new property right regime is accepted and enforceable at the local level.

Environmentally friendly technological alternatives to traditional technologies, if these alternatives can profitably compete with current practices, will likely be adopted by users without specific incentives that encourage their use or disincentives that discourage the use of environmentally more damaging technologies (Qaim et al. 2006). If, however, access to such technologies is limited by liquidity constraints, interventions such as rural credit schemes have been shown to increase adoption rates and improve ES provision levels (Anderson et al. 2002; Anderson and Thampapillai 1990).

⁴ For a list of reasons why the private sector will not provide the needed goods or services, see technical appendix in Belli et al. (2001).

Government and civil society engagement in environmental education and awareness building has shown to be a major contributor to reducing the costs of environmental management by affecting human behaviour in ways that narrow the gap between privately and socially optimal ES flows (Kollmuss and Agyeman 2002; Palmer et al. 1998).

In some cases, relatively small investments in establishing mutually beneficial partnerships can help solve environmental problems (Schwartzman and Zimmerman 2005). Research on partnerships, however, has also shown that implementing such enabling management instruments often requires long-term coordination and the establishment and maintenance of a legal regulating framework (Visseren-Hamakers and Glasbergen 2007), all of which can be expensive.

Finally, farm income (a commonly used indicator of human welfare) and some extreme ES flows are directly and negatively linked; for example, excessive rainfall can cause flooding that destroys crops or droughts can make agriculture infeasible. Insurance schemes have traditionally been used to mitigate the negative effects related to extreme weather events (Hazell et al. 1986), and the public sector has played important roles in establishing, monitoring and guaranteeing such schemes. In situations in which risk undermines the incentives to adopt ES-friendly technologies or farming practices, insurance schemes can contribute to stabilising or increasing incomes, and to ES conservation (Nail et al. 2007).

2.2.2 Incentive-Based Management Instruments

Whenever ES are underprovided, overused or underinvested in from a social perspective, one frequent cause is that the value of that ES is not evident to or captured by the individuals influencing their provision. In such situations, governments and local ES beneficiaries have often decided to provide direct incentives that encourage ES conservation or land use practices that provide additional ES (Portney and Stavins 2000). Subsidies represent one way of providing incentives, for example, by reducing the costs of fertilisers and fuel or providing cheap credit lines for particular agricultural activities (Huber et al. 1998; Lowe et al. 1999). However, subsidies supporting the production of goods that intensively use (e.g. water for agriculture) or compete with (e.g. forest clearing for agriculture) ES have often contributed to ES losses (Brouwer and Lowe 2000). Nonetheless, these policy instruments can be used and combined with other measures to change behaviour in ways that generate or protect ES (Oenema et al. 2006).

Payments⁵ are generally perceived to be the most direct way to stimulate the provision of a given ES, and while few concrete examples are currently in place,

⁵The 'environmental services' addressed by most existing PES schemes are equivalent to ecosystem services with public good character, for example, carbon fixation and biodiversity-related benefits, or scenic beauty (Landell-Mills and Porras 2002).

PES has received much attention in the recent environmental management literature (Börner et al. 2007; Milne and Niessen 2009; Wunder et al. 2008). Costa Rica was one of the first countries to implement a national PES scheme to manage ES, such as biodiversity conservation, soil erosion control, water flow regulation and forest carbon retention (Pagiola 2008). However, the cost-effectiveness and equity effects of these pioneering projects have yet to be comprehensively assessed, especially for the case of large-scale interventions (Pattanayak et al. 2010; Wunder 2008).

Certification or ecolabelling is a widespread management instrument used to increase the market prices of products as an incentive in favour of ES-friendly production practices (Ferraro et al. 2005). The certificate or label is used to separate markets for conventional and more eco-friendly products and to allow consumers the option of paying (at a premium) for the improved management of ES. Some authors therefore refer to certification also as a market creation management instrument (Nunes and Riyanto 2001). Establishing and managing certification and ecolabelling schemes can be expensive, and an array of product, market and other conditions must be fulfilled for certification schemes to be successful in developing country contexts (Ebeling and Yasué 2009).

2.2.3 Disincentive-Based ES Management Instruments

Disincentives are the most commonly used policy instruments for environmental management, especially in Latin America (Huber et al. 1998; Seroa da Motta et al. 1996). Whenever the costs associated with ES use or modification are perceived by society to be excessive, disincentives can be used to reduce and regulate the agricultural or other activities that are causing ES losses. Measures such as regulations (e.g. forest retention laws), bans (e.g. trade bans on endangered species) and standards (e.g. gender and size restrictions on the harvesting of certain types of wildlife) are typical examples for disincentive-based management. Compliance is rarely voluntary, so fines and legal action (e.g. confiscation or even imprisonment) are often used to enforce compliance (Pearce and Turner 1990). Disincentive-based management has been largely criticised as being economically inefficient and as having negative effects on poverty (Dietz et al. 2003; Holling and Meffe 1996). Recent evidence, however, suggests that enforcement of forest conservation law has significantly reduced deforestation in parts of the Brazilian Amazon (Hargrave and Kis-Katos 2010). Yet, independent of impact assessments, disincentive-based ES management has remained popular among public policymakers, in part because regulations are relatively easy to establish (though eventually costly to enforce, especially in the context of developing countries, Robinson et al. 2010) but also because they can generate government revenue in the form of fines.

Environmental taxes, for example, on land, represent another disincentive-based policy option that has been shown to bring about both ES *and* welfare gains at least in the context of developed countries (Bosquet 2000; Johnstone and Alavalapati 1998). In developing countries, however, levying taxes to enhance ES flows can

have adverse effects on the asset portfolios and income flows of the poor (Bruce and Ellis 1993).

User fees represent an option for managing ES use and modification at the local level, for example, in national parks and other forms of protected areas (Green and Donnelly 2003). User fees, for example, in the form of resource extraction permits for timber or non-timber forest products, also represent a form of regulating resource use and extraction (and hence some ES associated with these resources) (Simula et al. 2002; Sudirman and Nely 2005). Research has shown that large economic benefits can be derived from allowing controlled access to and use of protected areas, especially if they are successfully integrated in local and international markets for tourism and other rather ‘nondestructive’ uses (Amend et al. 2006). Whether protected areas achieve conservation objectives crucially depends on effective enforcement. Nonetheless, two global studies have recently confirmed that protected areas have worldwide significantly reduced the pressure on protected forests (Nelson and Chomitz 2011; Porter-Bolland et al. 2011). Surprisingly, community-managed forests and extractive reserves fared better than strictly protected areas in terms of conservation effectiveness, suggesting that protection need not be at odds with the sustainable management of forest resources.

2.3 Factors Affecting the Performance of ES Management Instruments

In this section, we discuss three sets of factors that influence the effectiveness of, and the co-benefits generated by, ES management instruments. We, first, focus on the biophysical characteristics of ES, such as spatial and temporal characteristics as well as complexity (Daily et al. 2009; Fisher et al. 2009; Kremen 2005). Second, we examine the socioeconomic conditions of natural resource users, which have recently become a key issue in the debate on the scaling up of incentive-based ecosystem service management under an international, REDD mechanism (Agrawal et al. 2011). Third, we explore the institutional environment for ES management that governs, at least in the short term, the pathways through which incentives, disincentives and enablement measures can be delivered on the ground and thus critically determines their cost-effectiveness (Howlett 2004; Ostrom 2008).

2.3.1 The Biophysical Characteristics of ES

The MA (2005) definition of ES included many types of benefits that humans can obtain from the environment. Understanding some of the basic properties of this mixture of ecosystem services and goods is therefore the first step to evaluating the potential costs and effectiveness of alternative management instruments.

2.3.1.1 Complexity and Interdependence

All ES result from complex geobiophysical interactions, but not all ES are equally 'systemic'. Promoting carbon sequestration, for example, requires far less knowledge about ecosystem functioning than enhancing soil fertility or species diversity, both of which depend much more on the presence of a variety of ecological processes, and hence may be more vulnerable to ecosystem modifications. This biophysical 'independence' and (hence) predictability of management-instrument-specific effects has made it easier to develop quantitative measures of carbon stocks and flows in a variety of land use systems and to identify and test policy approaches to managing this important ES in the context of forest and in agroecosystems (Fisher et al. 2009; Pagiola et al. 2002). Ecosystem services related to biodiversity, on the other hand, tend to be more complex and interdependent, for example, the number of trees in a certain environment can be as important as the composition of tree species for habitat quality (Arroyo-Rodríguez and Mandujano 2006).

The MEA classifies ES based on their key functions in an ecosystem into regulating, provisioning and supporting (among others) services. This classification approach is particularly useful to identify the amounts and pathways through which ES benefits flow to specific stakeholder groups. Knowing these pathways facilitates the process of determining who could and should cover at least some of the costs of managing ES. From a management point of view, however, this categorisation of ES may be less convenient, because it groups ES with very different characteristics in the same categories. For example, managing flood versus climate-regulating ES requires hugely different sets of knowledge and policy instruments. Identifying cost-effective intervention mechanism thus often requires better knowledge about the implications of specific ES characteristics for management (Kroeger and Casey 2007).

2.3.1.2 Non-excludability

Both the challenge and the need for managing ES arise from the fact that the benefits derived from them are often 'non-excludable'. Consider, for example, carbon sequestration in forest plantations: the climate-regulating functions of these plantations accrue to the society as a whole and not exclusively to the individual even if the individual owns the land on which the carbon was sequestered. Soil quality on private land, on the other hand, is an excludible ES that provides benefits exclusively to the owner. Conserving soil quality is thus often in the best interest of the owner.

If all benefits of a given ES can potentially be captured by the stakeholders that affect their provision, incentives and disincentives intended to increase their provision levels will usually not remove the underlying causes of ES losses. In the case of soil quality, the causes of degradation can often be traced back to the lack of access to soil quality conserving agricultural practices (Vosti and Reardon 1997).

2.3.1.3 Temporal and Spatial Dimensions and Interdependency of ES Provision

The provision of ecosystem services varies naturally over time and space (Kremen 2005). For example, rivers reach the ocean at specific points, and although the water they carry may influence ecosystems for many miles out to sea, eventually these influences disappear. Elevation patterns in mountainous regions, such as the Andes, introduce enormous spatial variations in the provision of ES, especially if they depend on climate conditions. But even in regions with less heterogeneous elevation patterns, such as the Amazon basin, bedrock characteristics and tidal inundation introduce additional spatial variability to ES provision and related benefits. For example, tidal movements in the Amazon allow for electricity generation in small-scale tidal power plants along some, but not all, rivers and temporally inundated areas (Charlier 2003).

Often there are also multiple temporal patterns to ES provision – diurnal, monthly, seasonal and interannual. For example, huge diurnal temperature variations in some mountainous regions affect the types of vegetation that will naturally occur and the agricultural crops that can be grown. Or, to take another example, river discharge in the Amazon has been shown to vary enormously depending on the ENSO cycle with implications for hydropower generation, fluvial transport and fishery production (Richey et al. 1989). These natural patterns affect the patterns of responses of ES to management interventions.

With regard to both space *and* time, there can be great uncertainty regarding ES flows. For example, we may not know where the end point of influence of a particular stream flow might be at a given point in time, because weather patterns in a given year can extend or reduce that stream's flow. Some of this uncertainty can be reduced or better understood with the proper investments in research/monitoring, but other aspects of this uncertainty will be difficult to discover; this is especially true for ES that have yet to be concretely defined or measured. Regardless, decision-makers wrestling with selecting management instruments and crafting them to be cost-effective in specific agroecological and socioeconomic circumstances should know that while 'on average' these management instruments may succeed in meeting stated ES objectives, there will be times and places (within their target temporal and geographic domains) when/where their final mode of intervention will overshoot and undershoot these same objectives. The costs to society of these over- or undershootings may be quite significant.⁶

Finally, as a result of these spatial and temporal dynamics, the social benefits of ES management in a given place and time may accrue to different stakeholder groups many years later, for example, slowing deforestation in the Amazon may help retain historical rainfall patterns at a continental scale (Werth and Avissar 2002).

⁶ For example, the value of surface water during the wet season can be much lower than the value of surface water during the dry season (Torres et al. 2012).

2.3.1.4 Implications of ES Characteristics for Choosing ES Management Instruments

Complexity, interdependence and uncertainty in temporal and spatial ES dynamics mean that management interventions can be ineffective or, worse, produce unexpected negative consequences. An important first step toward managing ES is to recognise that humans simultaneously adapt to and change ES provision through activities that alter natural temporal and spatial dynamics. These activities and related investments are undertaken to harness the private benefits of ecosystem services (e.g. diverting river flows to irrigate agricultural products) or to reduce the private damages associated with ecosystem disservices (e.g. building levees to reduce flooding), or for reasons that are not directly related to ES but still can affect ES flows. ES-modifying activities are thus not distributed randomly over the landscape; spatial patterns of investments and activities will be carried out where their private economic returns are positive, and they will be undertaken first in areas where these returns are highest (Chomitz and Thomas 2003; Pfaff 1999; Thünen 1826). Rapid and unexpected changes in ES caused by management interventions thus imply potentially large costs to those who are particularly dependent on (or who have adapted to) the ES targeted by policymakers for intervention. This often applies, for example, to downstream water users after the construction of dams.

It should also be noted that changes in natural phenomena can have similar welfare-reducing effects on individuals and communities that have developed livelihood strategies based on expected ES flows. For example, the 2005 drought in the Amazon exemplified the implications of water shortages to the local and regional economies, for example, disruptions in local and regional river-based transport, that have developed under conditions of relative water abundance (Zeng et al. 2008).

A first step toward ES management is therefore to understand how the characteristics of the targeted ES and its current use affect:

1. The types of benefits that the ES provide (e.g. income, air quality)
2. The ways in which these benefits are generated (e.g. income via agriculture)
3. The ways in which temporal and spatial patterns of benefits are generated
4. To whom these benefits directly and indirectly accrue

The second step is to ask how alternative ES management instruments affect these four items.

In practice, we may not have the means to measure actual ES, for example, accurately measuring watershed services can be exceedingly complex and expensive (Perrot-Maitre 2006). Hence, decision-makers are often forced to manage land uses (or other broader units that are relevant for ES provision) in the hopes of influencing specific ES flows (Wunder 2005). A common approach to managing these cases involves identifying (but not necessarily measuring) the target ES and the land uses or land cover types deemed most likely to generate it (e.g. river bank vegetation to reduce erosion and thus water turbidity levels). It is assumed that if the policies are successful in retaining or expanding the target land uses, this success will be proportionately replicated for the targeted ES. The literature has pointed out the problem

associated with many of the assumptions underlying this approach, for example, the extent to which heterogeneity within land use categories can affect ES flows. For example, a forest use regulation and a cap-and-trade system may be equally effective in retaining a specific total amount of forested land; however, the geographic distribution of the forests retained by the two interventions will likely be different, with possible consequences for some ES (Debinski and Holt 2000).

One important implication is that decision-makers are generally forced to manage ‘bundles’ of ES by selecting management instruments that affect land use and land use change. Bundling may often allow decision-makers to credibly suggest that unknown or highly undervalued ES are included in these bundles (e.g. conserving primary forest carbon stocks through REDD will also conserve biodiversity). But, depending on how well individual components of a ‘bundle’ can be measured, this approach makes it harder (and more expensive) to identify the beneficiaries of ES management actions and to articulate demand for management actions (Wunder and Wertz-Kanounnikoff 2009).

2.3.2 Institutional and Socioeconomic Factors Affecting the Performance of ES Management Instruments

A series of institutional factors can affect the performance of ES management instruments. Often, ES benefits are not directly related to natural characteristics. Instead, various layers of property rights attached to natural resources through legal and/or customary norms and regulations usually govern local access to and use of ES, for example, living next to a river does not necessarily convey water use rights (Ostrom et al. 2005; Schlager and Ostrom 1992). In developing country contexts, the enforcement of legally defined property rights is often weak. In the Amazon and in the neighbouring Andes region, many natural resources are de facto open access resources with ill-defined, incomplete, nonexistent, conflicting or weakly enforced property rights (Ravnborg and Guerrero 1999; Seroa da Motta and Ferraz do Amaral 2000).

Lack of administrative capacities and operational infrastructure is often the reason for poor enforcement of property rights and at the same time limit the effectiveness and increase the costs of incentive and disincentive-based management options (Börner et al. 2011; Robinson et al. 2010). For example, PES schemes may be ineffective if the recipients (e.g. landowners) cannot exclude others from modifying ES originating from their land. Moreover, if property rights are poorly defined, regulators will find it harder to establish liability for illegal natural resource degradation. Some research, nonetheless, suggests that offering PES may actually encourage property right enforcement by rural communities and, overall, lead to positive environmental and welfare outcomes (Engel and Palmer 2008). Effective property right transfers or supporting local communities to build and maintain efficient local institutional arrangements that regulate resource use and access will nonetheless often be necessary to address situations where property rights are ill defined and/or weakly enforced (IFAD 2003; McGrath et al. 1993).

Even if effective institutions are in place to implement and monitor ES management instruments, socioeconomic conditions will play an important role in affecting the performance of ES management instruments. A straightforward rule that applies to incentives (e.g. payments) or disincentives (e.g. fines) for ES management is that the costs of noncompliance must outweigh the benefits, that is, compliance levels tend to be low if the expected value of fines are smaller than the expected benefits associated with noncompliance, because of low fine levels and/or low probability of enforcement (Becker 1968). Imperfect enforcement of payment contracts may have even worse implications for the cost-effectiveness of this particular management instrument, because higher (than opportunity cost) transfers will be needed to pursue recipients to comply with the conditions attached to payments (Börner et al. 2011).

Poverty, which is often associated with (or caused by) limited access to basic public services, credit and agricultural technologies, can represent a significant barrier to cost-effective ES management. If disincentive-based interventions restrict poor people's access to essential natural resources and their locally valued ES, management instruments may be ineffective, at best, or even deepen poverty. Poverty also typically coincides with poor institutional and organisational capacity, which represents a challenge for most enabling and incentive-based intervention options. It has, for example, been argued that the spatial coincidence of poverty and valuable ecosystem services, especially in Latin America, comes with an opportunity for achieving win-win outcomes through PES. Several studies, however, emphasise that high transaction costs (for both scheme implementers and for the poor) and the lack of formal land titles can limit the participation of poor ES service providers in such conditional incentive schemes (Pagiola et al. 2005; Pfaff et al. 2007; Rios and Pagiola 2010). Poverty alleviation objectives of PES, in addition, may be jeopardised if landless poor rural dwellers lose employment opportunities in set-aside (purely conservation-oriented) as opposed to asset-building (e.g. reforestation) schemes (Wunder 2008; Zilberman et al. 2008).

Farmers may also not be poor by standard poverty measures, but still be too poor to invest in sustainable land use practices, that is, they may suffer from conservation investment poverty (Vosti and Reardon 1997). In such cases, disincentive measures intended to induce more intensive land uses may fail to bring about the desired outcomes. However, even PES may fail if conservation investments are subject to severe liquidity constraints, for example, the purchase or rental of heavy land machinery. Yet, depending on conditioning factors, poverty may also inhibit environmental degradation. For example, subsidies to encourage investments in seemingly sustainable land uses can result in more, rather than less, deforestation and declines in forest ES (Börner et al. 2007; Vosti et al. 1997).

Finally, many management instruments in the enabling category, such as community-based resource management, require collective action, civil engagement and local organisational capacity, result in enhanced ES provision (Kellert et al. 2000). In recently settled areas, such as at the margins of tropical moist forest, most of these types of social capital are often in short supply (Fearnside 2001). Interventions that rely heavily on preexisting social capital may thus often represent promising options only in the long run and face large establishment and maintenance costs.

2.3.2.1 Implications of Institutional and Socioeconomic Factors for ES Management

Local institutional and socioeconomic circumstances mean that no blueprint approaches exist to managing ES, especially in the developing world. A couple of general lessons nonetheless emerge. First, when ES losses are the result of lacking formal or customary institutional structures and/or poverty, direct ES management through most incentive and disincentive-based instruments is unlikely to remove the root causes of ES loss and be ineffective and/or costly at best or result in undesired outcomes at worst. Enabling measures, such as provision of basic education and other public services as well as improving access to locally adapted technological innovations may instead often represent more effective initial investments toward improving ES provision.

Second, the size and timing of the net benefits associated with ES-modifying activities and investments and the stakeholder groups to whom these benefits accrue will influence ES management outcomes. For example, where such activities and investments are very profitable (as is often the case for the extraction of precious timber resources), PES schemes will be expensive and perhaps beyond the fiscal means of policymakers or willingness of ES beneficiaries to pay. Under such circumstances, land use taxes (which reverse the financial flows among stakeholder groups, vis-à-vis PES) or land use regulations may be more cost-effective ES management instruments, even when enforcement/monitoring costs are included in the comparison.

Third, at least in the short term, trade-off relationships between ES objectives and other development objectives are the rule rather than the exception.⁷ Hence, ES managers must recognise that individual ES management instruments may often not achieve both poverty alleviation and ES management objectives. Poverty alleviation generally requires a broader development approach involving various non-environmental policy investments and activities. The existence of trade-offs also often calls for negotiation-based solutions, in which stakeholders need equal opportunities to guarantee fair outcomes. If direct negotiation between ES users and modifiers is an option, for example, in small watersheds, PES schemes may often require little or no government involvement. Reducing deforestation at a larger scale will, however, always depend on the involvement of governments, because ES beneficiaries lack the means and legal mandate to monitor and enforce conservation contracts.

Fourth, because ES management is often land use based, rural landless people may be affected in unintended ways. Especially when ES management options, such as REDD, are applied. ES managers must therefore consider safeguards for this and other vulnerable groups.

⁷ More fundamentally, economic efficiency requires identifying specific policy instruments to resolve specific policy problems; it will rarely be the case that an environmental policy instrument is the most efficient way to resolve (say) an economic development problem.

Finally, an old paradigm is that local problems require local solutions. As we have seen, this holds for some but certainly not all problems related to ES provision and use. Many central governments have tried to address this notion by delegating the management of some natural resources (and hence ES) to lower-level administrative units, such as states or districts, in both developed and developing countries. Decentralised management, however, poses new challenges to effective and efficient ES management, among them the risk that unprepared and underfinanced local governments lack the administrative and technical capacities to take and effectively implement policies related to ES flows (see Toni and Kaimowitz 2003 for the example of forest management in the Amazon).

But even if local governments and local civil society *are* prepared to cost-effectively handle local ES flow management challenges, such challenges are generally not exclusively local but rather ‘spill over’ spatially and temporally into the domains of other decision-makers. Some very important ES flows with large public good components do not coincide with, or are not contained within, administrative boundaries, so managing them requires cooperation across policymaking boundaries. Among the Amazon and Andean countries, this notion has given rise to the foundation of the Amazon Cooperation Treaty Organization (ACTO). However, multilateral environmental agreements and partnerships around the world are plagued with the same difficulties, for example, free riding (individuals, communities or even countries reaping the benefits of ES management without paying their share of management costs) and high transaction costs of intergovernmental negotiations (Chang and Rajan 2001).

2.4 ES Management Instruments and Expected Performance: An Overview

This section highlights selected factors that our review of the literature suggests are likely to affect cost-effectiveness and poverty alleviation objectives of ES management. Table 2.1 summarises key factors that (if present) influence the performance of selected ES management instruments.

Column 1 of Table 2.1 identifies the management option type, column 2 identifies the ES to be managed, column 3 addresses poverty alleviation, column 4 identifies factors that affect the cost-effectiveness of reducing poverty and the final column (5) examines factors influencing the cost-effectiveness of achieving ES management objectives.

Beginning with column 2, it is clear that most instruments can be used in theory to address either specific ES *or* a bundled set of ES. In practice, however, the great majority of ES management instruments has been used to influence human behaviour with respect to broad natural resource categories, such as forests or fishing grounds among many others (as suggested in Fig. 2.2), with expected direct effects on specific ES flows, most of which are not measured in detail. The notable exception

Table 2.1 ES management instruments and factors expected to influence their cost-effectiveness

ES management instrument	ES target	Conditions with potential to reduce (or not exacerbate) poverty	Factors reducing cost-effectiveness in achieving poverty alleviation if present	Factors reducing to cost-effectiveness in achieving ES objectives if present
PES (subsidies)	Specific Carbon Biodiversity Water Scenic beauty	If the poor can offer additional ES provision and If the poor have rights to exclusion	Poor can offer little additionality Poor have weak and insecure property rights No legal basis for PES from treasury	Spatial heterogeneity of opportunity costs High opportunity costs High transaction costs (monitoring and enforcement) Weak enforcement capacity High transaction costs (monitoring and enforcement)
Taxes (user fees, etc.)	Specific and unspecific Timber NTFP Water Carbon Biodiversity Scenic beauty Climate regulation	If tax revenues are reinvested in (compensating) poverty alleviation measures or If the poor can directly capture the revenues or If the poor are excluded from the tax	If poor are allowed to capture user fees, they often lack capacity to attract users Few examples of environmental taxes	
Certification	Specific and unspecific Water/air quality Carbon Biodiversity Soil quality Climate regulation	If the poor have market access and If the poor can capture price premiums and If the poor can meet quality standards	Limited market access Costs of certification process Costs of investments in to technology	High transaction costs (monitoring of difficult to measure ES) Weak enforcement capacity

Technological innovation	Specific and unspecific Various	If the poor have technology access (knowledge, extension, infrastructure) and If the poor can afford up-front investment costs	Limited technology access Investment poverty	High opportunity costs of conservation investments High development costs (R&D system)
Regulation (bans, standards, protected areas)	Specific and unspecific Various	If regulations do not affect essential ES consumption (or if so, compensating mechanisms are in place) or If regulations include special treatment of poor ES users	Low bargaining power to negotiate compensation and special treatment	Weak enforcement capacity High transaction costs (monitoring and enforcement) if under pressure
Local institutional arrangements (community-based resource management, partnerships)	Specific and unspecific Various	If the poor have equal bargaining power in the negotiation process or If institutions are set up in a way that promotes poverty alleviation	Low administrative and organisational capacity of local governments Low bargaining power Few cooperative experiences	High returns to free riding
Environmental education	Specific and unspecific Various	If the poor have access to education (infrastructure, costs) and If education addresses ES issues relevant for the poor	Low level of teacher training in rural schools High costs due to poor transport infrastructure	High returns to environmentally damaging behaviour (lack of rule of law)
Cap-and-trade schemes	Specific Air quality Climate regulation Various others	If the poor are sellers and If the poor have market access	Limited market access	Weak enforcement capacity High transaction costs (monitoring and enforcement)

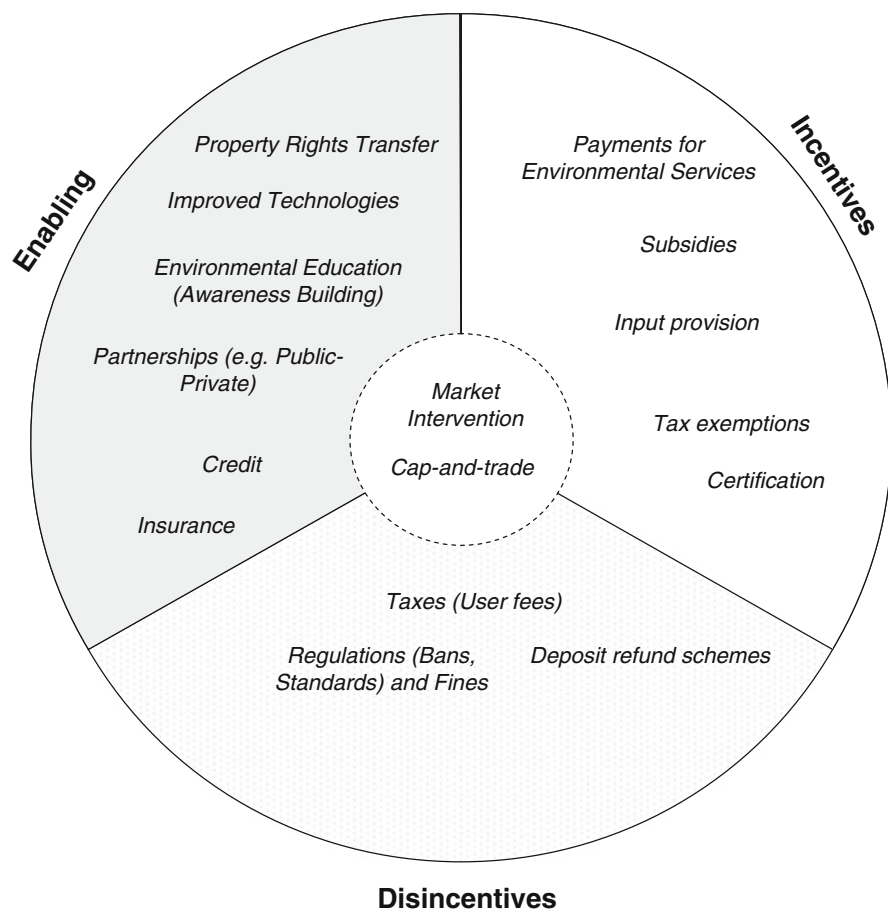


Fig. 2.2 ES management instruments by means of impact

are PES schemes that are often specifically designed to address one or two well-defined ES, such as carbon or watershed services, but which likely have spillover effects (of different magnitudes and perhaps in different directions, vis-à-vis the targeted ES) on other ES.

Almost all instruments can potentially be designed in ways that leave the poor unaffected, or perhaps even better off (column 3 in Table 2.1). However, designing and implementing measures to achieve poverty neutrality or to reduce poverty generally implies additional up-front costs (e.g. those associated with building participatory or institutional capacity) and operational costs, and decision-makers have not always been willing or able to incur these costs. Poverty effects, nonetheless, tend to vary across ES management instrument categories. Poverty effects of

‘enabling’ instruments, for example, those delivering technological innovations, tend to depend on whether the poor will be able to reap their benefits of ES management. When access to technological innovations is limited, the poor generally do not benefit and may even become poorer, for example, when productivity increases among nonpoor adopters result in lower product prices. Or, the poor may lack experience and (hence) skill and bargaining power, which can limit their ability to effectively participate in the design of ES management, such as community-based resource management or public-private partnership agreements.

In the case of disincentive-based instruments (e.g. taxes or fines), mechanisms need to be in place to compensate low-income groups (e.g. tax exemptions) for negative welfare consequences or instruments need to be designed in ways that leave the poor unaffected (e.g. allowing the resource-poor to continue to engage in subsistence activities in protected areas). Incentive-based instruments, such a PES, on the other hand, often require a minimum level of market access to work effectively. For example, conservation payments to farmers living in remote areas without access to food markets may not compensate for losses in subsistence production. Also, mechanisms need to be in place to make sure that price premiums actually trickle down to the poor instead of being captured by intermediaries, as has been the case for some forest certification schemes (Harris et al. 2001).

In many countries, ES modifying activities are already highly regulated, at least on paper. For example, over 40% of the Brazilian Amazon region is covered by various categories of protected areas and indigenous territories, whereas the remaining public and private land is subject to the national forest retention standard that requires 80% of landholdings to remain under primary forests. Many other countries, like Peru, have banned deforestation almost completely. In practice, however, deforestation continues to take place (illegally) wherever it is profitable to do so, that is, mainly alongside roads and highways (Laurance et al. 2002). Increasing the effectiveness of existing regulatory policies by enforcing them more rigorously is thus often seen as a low-hanging fruit for ES management. But, especially in remote parts of the Amazon, where field-based enforcement of disincentives can operationally become more costly than providing incentives, PES for avoiding deforestation may come to be a cost-effective complementary measure (Börner et al. 2010; Nepstad et al. 2007; Swallow et al. 2007).

Where property rights are secure, resource users are relatively homogeneous and communities are willing to cooperate; building capacities for more effective ES management is likely to help maintain essential ES and contribute to poverty alleviation. However, strong incentives for ES modifying activities will always represent a major challenge for ES management; wherever such incentives are high, ES management will also be costly. This is especially true if ES do not provide direct benefits to resource users or if beneficiaries do not have a voice to negotiate ES outcomes. Hard trade-offs therefore need to be faced by those that promote economic growth and infrastructure development, for example, at forest frontiers, unless the global community that benefits from forest-based ES is willing to compensate land users for foregoing economic opportunities.

2.5 Implications for Research, Capacity Strengthening and Policy

This chapter reviewed the theoretical and applied literatures on instruments that can be or have been used to manage ecosystem services (ES). The primary objective was to explore the biophysical and socioeconomic factors that affect the cost-effectiveness of alternative ES management instruments. A second objective was to assess the effects of ES management instruments on the welfare of the rural poor inhabiting areas targeted for ES management. We conclude here by indentifying key knowledge gaps with regard to three essential questions that decision-makers need to answer before making informed decisions on ES management.

First, what do we know about ES dynamics and their relationship with poverty? In terms of measuring poverty (especially measured in terms of income), we are on solid theoretical and empirical grounds – we can measure poverty and poverty dynamics – so any major gaps in knowledge are primarily attributable to insufficient resources having been dedicated to identify the poor and measuring the depths and nature of their poverty. As regards ES dynamics, the scientific base is much weaker – with relatively few exceptions (e.g. biomass carbon measurement), science has yet to provide decision-makers with practical measures of ES that capture their most important spatial and temporal dynamics. Complexity and uncertainty, however, suggest that such measures may often not exist. In times of climate change, research on how ES management can optimally deal with persistent and/or changing uncertainty is therefore all the more necessary.

Second, if stakeholders are unhappy with current levels of ES provision, why and what are the nature and degrees of their displeasure? Answering these questions requires knowledge of the private and social benefits associated with ES, the costs associated with changes in these flows and how these benefits/costs vary across stakeholder groups. Progress on this front has recently been made, for example, through initiatives like The Economics of Ecosystem and Biodiversity (TEEB),⁸ but large gaps in knowledge remain particularly in developing countries. The most important of these gaps is the development of methods to measure the benefits (though not always necessarily the monetary value) associated with particular ES or bundles of ES, and how these benefits change as ES are modified. To address concrete decision-makers' needs, these methods must incorporate the site-specific patterns of ES flows and generate ranges of expected benefit flows that capture the inherent uncertainty associated with important ES and their values to society.

Third, if policymakers decide to take action, what should be done and by whom? Answers to this question must build on the site-specific responses to the previous two questions plus an understanding of the determinants of the behaviour that modifies the ES in question as well as its responsiveness to management intervention. Research on land use and land cover change has made significant progress in understanding household level decision-making, but many of the most pressing

⁸ www.teebweb.org

ES management problems will have to be resolved at the community or at more spatially and socioeconomically aggregate levels. To extract lessons learned from current and past environmental policy initiatives, more rigorous impact assessments are needed that lead us to understand how policy design and the local conditioning environment affect the performance of individual and combinations of ES management instruments. Often such evaluations will have to rely on methods that allow establishing credible counterfactuals for policy interventions, such as matching analyses (Ferraro and Pattanayak 2006).

Finally, ES management choices are not merely decisions of independent social welfare optimising principals but the result of political bargaining processes. Work on new ways of establishing and managing dialogue related to ES management among stakeholder groups is progressing; such dialogue is generally seen as a necessary condition for achieving successful and sustainable outcomes. However, large gaps remain in identifying the most efficient methods of establishing and managing multi-stakeholder interactions and in developing mechanisms to generate and deliver needed science-based information into such discussion settings.

As the debate around reducing emissions from deforestation and degradation (REDD) evolves, many proponents begin to realise that only incomplete answers exist to the three questions posed above. The resulting uncertainty about how REDD could and should be implemented has led to mounting opposition against the concept that threatens its successful implementation (Agrawal et al. 2011). Forward-looking, scenario-based policy research that openly deals with the uncertainties attached to costs as well as environmental and social impacts of ES management could provide crucial input for a constructive, outcome-oriented policy debate.

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