Economic instruments for nature conservation

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"*The labour of Nature is paid, not because she does much but because she does little. In proportion, as she becomes niggardly in her gifts, she exacts a greater price for her work. Where she is magnificently beneficent, she always works gratis*." (Ricardo, 1817)

1. Introduction

The degradation of natural systems worldwide requires an appropriate policy response. The Millennium Ecosystem Assessment showed degradation is progressing at a rapid pace, and argued that reduced flows of ecosystem services (especially regulatory and cultural services) are likely to affect adversely the welfare of future generations. The World Bank has also documented ongoing processes of ecosystem destruction, and couches its interpretation in terms of capital depletion. Capital comes in different forms, and economists often distinguish between man-made capital, human capital and natural capital. Ecosystems are a specific form of natural capital assets, and provide a host of services. They maintain a genetic library, preserve and regenerate soils, fix nitrogen and carbon, recycle nutrients, control floods, filter pollutants, pollinate crops, operate the hydrological cycles, etc. Degradation of ecosystems is much like the depreciation of physical capital (e.g., roads, buildings, machinery) but with two big differences: damages are frequently hard to reverse, and ecological processes tend to be non-linear, so that ecosystems can collapse abruptly, without much prior warning. Another important difference is that property rights over natural capital are often unclear, which substantially complicate the matching of costs and benefits within and across generations.

Empirical work demonstrates that aggregate *capital stocks* may be falling, even if *income* is growing (e.g., Dasgupta 2001). This may happen, for example, when forests are cut and the proceeds are used for consumption (rather than investment in alternative forms of capital, such as infrastructure). During the period 1970-2000 this was true for all Asian countries, except China. Africa was even worse off, with both income and capital stocks falling. Looking at national income as a proxy for welfare is misleading, as income growth financed by capital depletion ("selling the family silver") will be only a temporary reprieve. Capital stocks are a measure of true wealth, and are the basic source from which future income is derived. This is one of the main reasons why economists are interested in the

conservation of nature. In addition to the conventional "efficiency" concerns that are the bread-and-butter of economists, issues like inter-generational equity also come into play.

Conservation of nature might "pay" from an economic perspective: it may be rational to conserve natural capital, rather than to convert it into alternative forms of capital or to consume it today. Some forms of natural resource use leave natural habitats more or less intact – think of reduced-impact logging or sustainable fishing – while others convert natural systems into something completely different (plantation cropping, intensive farming, etc.). An overview paper that compares the profitability of various forms of land use (Balmford *et al*. 2001) identifies many examples where investing in nature pays: sustainable management of ecosystems yields a greater discounted flow of benefits than going for short-term gains (where discounting a flow of benefits implies carrying future benefits forward to the present, so that they are commensurate with current benefits and costs—for this purpose economists employ a discount factor).[1]

To observe that nature conservation pays does not imply there are no incentives for resource managers to destroy it (or to convert it into other forms of capital). An important reason why nature may be destroyed is that many of the valuable services and benefits nature provides are "free" in the sense that they are not marketed. This is an obvious case of *market failure*. Even if nature is valuable, as long as it does not command a flow of money, its value tends to be ignored by private parties. Examples include the air purifying qualities of ecosystems, but also non-use values associated with biodiversity conservation. Environmental economists have developed methods to attach monetary values to ecosystem services, but that is not the same as money-in-the-bank. The point is that even if the *social*

[1] Of course counter-examples also exist; sometimes converting natural capital to financial capital provides higher returns. This is particularly probable when natural resource stocks are plentiful, and when the perspective of private investors or resource owners is adopted (see below).

returns to conservation and sustainable management are high, from the perspective of a *private* individual it may be better to ignore some of these benefits (i.e., those accruing to others) and focus on the small subset of tradable benefits only. Hence the conversion of nature into alternative forms of capital may be privately optimal and socially non-optimal at the same time. Examples include farmers who have incentives to intensively manage their lands at the detriment of the conservation of meadow birds, or whalers who believe that whales are more valuable in the bank than in the water.

Market failure implies that governments and other intermediaries can play a role to improve social welfare—the sum of payoffs for producers and consumers (citizens). Since market failure occurs at multiple scales – from the local to the global level – the appropriate intermediary varies from case to case. However, history has taught us there are many cases where public policies aggravate problems (or even create them). Examples of *policy failure* include "perverse subsidies" – subsidies that are bad both for the economy and bad for nature. The total amount of perverse subsidies, in both developed in developing countries, may be as high as US$2 trillion per year (Myers and Kent, 2001). A vivid example of a perverse subsidy relevant to biodiversity conservation would be investment subsidies and tax holidays for fishers, or subsidies to convert tropical forests into rangeland.

Nevertheless, it is understood that the government has a role to play in promoting nature conservation. In the past, two approaches to conservation have been extensively tried: traditional 'command-and-control' approaches emphasizing protected areas (bordered by fences or otherwise) and other disincentives to degradation, and an indirect approach called 'integrated conservation and development projects' (ICDP). ICDPs try to reconcile community development and conservation by promoting sustainable resource use or alternative sources of livelihood (which explains why ICDPs are occasionally referred to as 'conservation by distraction'). Both approaches have yielded some positive effects; but it is

becoming increasingly clear that a third wave of more direct, targeted incentives is needed to make further progress on the conservation and development front (Ferraro and Kiss 2002).

Increasingly, policy makers turn to economists for advice on how to design efficient and effective conservation policies. Cap-and-trade programs are one solution, combining command and control with market mechanisms. This only works if the regulator is able to set the cap and has the power to force producers (traders) to play according to the rules. People now trade fish quota the world over, trade the right to emit sulphur in the USA, and trade the right to emit carbon in Europe. Often markets for trading did not exist; they had to be created.

Consider the example of fishing. People have been trading fish for a long time, but trading the right to go out and catch fish is more novel. How does this work? Policy makers have to put an upper limit (or "cap") on the quantity of fish that may be harvested. These harvest rights are transferred to the fishermen – through any of a variety of allocation mechanisms – who are then allowed to trade amongst themselves, thereby determining where the value of the rights is highest, i.e., who can harvest at lowest cost. Through trade, firms themselves decide who will harvest, this is not done by relatively uninformed regulators (who are only enforcing aggregate harvesting, so that the cap is not exceeded). Trading harvest quota implies maximum flexibility in choosing the distributional implications of management, and also provides incentives to improve technologies.

Subsidy or tax systems are alternative market-based instruments. One important difference between cap-and-trade systems versus tax-and-subsidy schemes is that the former typically implies redistribution of rents within the sector (if rights are "grandfathered" or simply given to existing producers), while taxes imply a transfer of rents between sectors, e.g., from producers or consumers to the government. A third market-based instrument is payments for ecosystem services (PES).

In this chapter we describe various well-known conventional and market-based instruments for nature conservation. In the light of the recent attention for payments for ecosystem services as a tool to both promote conservation and alleviate poverty, we will pay special attention to this instrument. As will become evident, all instruments have advantages and disadvantages. However, intelligent application of economic instruments is likely to be a pre-condition for efficient and effective conservation of valuable natural capital.

2. Economic growth, poverty reduction and conservation

The relationship between economic growth, or poverty reduction more specifically, and environmental protection has long been central to economic debates about nature conservation. As economies grow, do increasing pressures on natural resources necessitate the use of economic instruments to protect the environment? Or will conservation emerge endogenously as a by-product of increased incomes and well-being?

For many years, roughly from the dawn of the modern environmental movement until the 1980s, most scholars and policymakers perceived a fundamental trade-off between environmental protection and economic growth. Conservationists commonly opposed economic development efforts and pushed for measures to address the environmental consequences of economic growth, in part by restricting or taxing resource use. Thus the gazetting of parks and protected areas was perhaps the central policy instrument for at least a century from the establishment of the world's first park, Yellowstone in the United States, in 1872.

By the mid-1980s, however, some evidence and theory had built behind the notion of synergies between environmental protection and economic development. As reflected in the well-known 1987 Report of the World Commission on Environment and Development – the so-called Brundtland Report – the prior belief in an intrinsic tension between growth and

development gave way to a more hopeful belief that technological progress could stimulate productivity, growth and thus economic development while simultaneously reducing pressure on natural resources. This hopeful view gained support as researchers recognized the strong correlation between areas with high rates of endemism in fauna or flora and high levels of poverty (Fisher and Treg 2007). The geographic co-location of efforts to reduce poverty through economic development and to conserve biodiversity naturally fed hypotheses that poverty and environmental degradation have common drivers that, if effectively addressed, could yield "win-win" results on both fronts simultaneously. Indeed, appropriate alignment of environmental protection and poverty reduction goals has long been a concern of many in the conservation community (McDonald et al. 2006, 2010).

This was the spirit behind the Convention on Biological Diversity (CBD), an international treaty agreed in 1993 that formally declared the conservation of biological diversity "a common concern of humankind" and an integral part of the development process. The prospect of discovering commercially valuable biochemical substances through "bioprospecting" is one of the many values the CBD associates with biodiversity conservation. Some early bioprospecting cases had been highly contentious, perhaps most notoriously the pharmaceutical multinational Eli Lilly's use of compounds found in Madagascar's rosy periwinkle to develop blockbuster cancer treatments, without compensation to the original source nation, one of the world's poorest.[2] Then in 1991 Merck reached an agreement that gave Costa Rica's National Institute for Biodiversity US$1 million and undisclosed royalties for any useful products derived from bioprospecting samples. The CBD then enshrined this principle of sharing of benefits between bioprospectors and source countries.

[2] Dwyer (2008) describes this and several other alleged cases of "biopiracy" in which multinational firms tapped indigenous ethnobotanical knowledge to generate commercially profitable products with little or no compensation to the communities from which insights and plant material were originally extracted.

Bioprospecting has largely failed to deliver on its promise. While biodiversity is valuable in aggregate, the economic value of the marginal species and of habitat conservation associated with the marginal species is likely low. Either there are multiple species that can produce the same commercially valuable compound, rendering the marginal value of any one source species low, or the probability of finding a unique source species is very low.[3] In either case, the expected profitability of bioprospecting, and thus the payments one could reasonably expect firms to make in order to help conserve biodiversity are likewise low (Simpson *et al*. 1996, Barrett and Lybbert 2000, Costello and Ward 2006). It is little surprise to economists, therefore, that bioprospecting has never generated significant funds for conservation activities nor tangible poverty reduction benefits.

So-called integrated conservation and development projects (ICDPs) temporarily flourished in the 1990s for reasons similar to those that fuelled enthusiasm for bioprospecting. The essence of an ICDP is the merging of conservation with poverty reduction and rural development goals, commonly advanced by attempting to promote livelihoods compatible with sustainable resource use in – or more typically around – parks and protected areas or by providing compensatory transfers to rural residents who use resources sustainably (Brandon and Wells 1992). The core idea was that by giving local communities a stake in maintaining biodiversity, the prospects for conservation success would increase relative to approaches that relied on government – or some other external agent – imposing resource use constraints. The problem is that ICDP designs – and community-based conservation designs more broadly – have commonly been based on untested assumptions about both human and ecosystem response (Brandon and Wells 1992, Barrett and Arcese 1995). As a consequence, common designs can actually hasten, rather

[3] In addition, many species are native to various tropical countries at the same time, so that these countries would have to act as a cartel to be able to reap the bulk of the benefits.

than avert, ecosystem collapse if the increased demand for natural products induced by local income growth outpaces the disincentive effects built into the quasi-contract between the conservation agency managing the ICDP and the local community, as Barrett and Arcese (1998) demonstrate using the example of the Serengeti Regional Conservation Strategy that introduced an ICDP intended to reduce poaching and thereby to reduce pressure on wild ungulates in northern Tanzania.

In spite of their obvious appeal, efforts to achieve "win-win" solutions – such as bioprospecting or ICDPs – have rarely fully delivered on their promise. The general problem, well-known to economists as the Tinbergen Principle, is that typically each policy objective requires its own policy instrument. Thus trying to achieve both conservation and development goals simultaneously through a single instrument is difficult at best. Economists typically think of "win-win" objectives as too lofty. Our more modest criterion for improvement, Pareto improvement, only requires making at least one person better off without making any worse off. The Pareto criterion implies that one should consider as a success any effort that achieves biodiversity conservation without imposing suffering on the poor, or sustainable improvement in living standards without compromising ecosystem function (Barrett *et al.* 2011).

A parallel literature attempts to reconcile disparate observations that suggest economic development and environmental protection are sometimes complementary and sometimes antithetical processes. The Environmental Kuznets Curve (EKC) hypothesis posits an inverted U-shaped relationship between various measures of environmental degradation and per capita income. Under the EKC hypothesis, economic growth in poor societies is initially associated with environmental degradation associated with increased effluent and waste discharge, habitat loss, etc. but as incomes grow further, beyond some turning point, economic growth begins to drive environmental improvement as pressures to

produce more natural resource-based commodities decrease and society comes to value and protect nature. The idea is that income growth brings greater capacity and willingness to pay for environmental protection, as well as technological advances that are less wasteful in resource use. After some unspecified threshold level of income these pro-conservation effects start dominating the environmentally damaging effects of increased resource consumption and waste generation. The powerful implication of the EKC is that while economic growth may temporarily despoil the environment, development is not a long-term threat: it is complementary to, or even necessary for, long term conservation success.

The empirical evidence in support of the EKC is, however, at best mixed. Most studies that support the EKC focus on measures of pollution (Grossman and Krueger 1995, Barbier 1997), although even those findings are contested (Harbaugh *et al*. 2002, Deacon and Norman 2006). Other studies focused on deforestation, energy use and other indicators typically find empirical little support for the EKC hypothesis (Koop and Tole 1999, Dasgupta *et al*. 2002, Mills and Waite 2009).

In the end, no compelling theoretical or empirical case exists for either universal trade-offs or synergies between economic and environmental goals. Complementarities between these desired outcomes appear feasible in many cases, but are too rarely achieved. The key determinants of successful synergy have proved rather elusive to identify. Because little observational or experimental data exist to describe and to test hypotheses about interactions in closely coupled human and natural systems in developing countries, we have only a sparse set of convincing empirical studies, which collectively suggest tremendous site-specificity of results, as well as dependence on the environmental problem at hand. Few truly generalizable lessons have appeared and associated policy debates often exhibit an ideological tone. The empirical evidence base and the theoretical literature underlying competing claims of trade-offs or synergies among economic and environmental goals yield

few unambiguous, generalizable conclusions (Lee and Barrett 2001). Perhaps the best guidance we economists can offer at this juncture is that one should anticipate trade-offs but work for synergies and be content to make advances in one dimension without worsening conditions in the other. We cannot be confident that nature conservation will "spontaneously" emerge as economies develop.

3. Regulatory approaches

Environmental economists distinguish between economic instruments for environmental conservation – manipulating incentives by altering relative prices — and so-called "command and control" approaches. Textbook economics tend to view command and control with suspicion. In the domains of pollution abatement and resource extraction, it is well-established that command and control is inefficient (i.e., does not achieve its objectives at lowest cost, partly because of information asymmetries between regulator and private parties) and fails to provide appropriate incentives for technological development (Perman *et al*. 2003). Nevertheless, while economic instruments such as taxes, subsidies and tradable permits are widely employed in pollution control and resource harvesting, command and control style regulation is still the dominant strategy in the domain of nature conservation.[4]

Global biodiversity conservation efforts rely heavily on protected areas, such as parks and reserves (MEA, 2005). In 1965, there were approximately 2,000,000 km^2 of national terrestrial protected areas. Forty years later, there were approximately 14,000,000 km^2, as well as millions more square kilometres of local, regional, indigenous and private protected areas. In total, more than 13% of the terrestrial surface of the earth is formally

[4] Note that tradable permits are a combination of command and control and economic instruments—before trade can start, aggregate harvesting (or pollution) should be "capped."

protected, and almost 4,000,000 km^2 more are managed as marine protected areas (IUCN and UNEP-WCMC, 2009).

The theory of protected areas is simple: if governments legally restrict human access to an ecosystem or its components, they decrease anthropogenic pressures, such as conversion and hunting, that otherwise would have taken place. Despite the simple theory, protected areas may fail to generate as much *avoided* pressure as anticipated. First, governments may be unwilling or unable to enforce compliance with protected area restrictions (so-called "paper parks"). The African "bushmeat crisis," in which illegal hunting in protected forests rivals habitat loss as a threat to endangered animal species, is a good example of legal protection failing to stem human pressures. Second, land users may recognize the correlation between regulations and the presence of intact habitat and wildlife populations, and thus pre-emptively destroy habitat and wildlife to reduce the probability of future regulations (Lueck and Michael 2003). Third, protection may displace pressures as demand for the protected ecosystem products (e.g., timber, crops, fish/meat) induces spatial shifts in production to unprotected locations.

Fourth, protection is often assigned to low-pressure areas, whose ecological trajectory may be little affected by the assignment of protection. Protected areas are often assigned to low-pressure areas because such assignment is politically expedient: influential citizens and firms do not protest the protection of areas that are of little productive use to them. For example, Andam *et al*. (2008) found that, in the well-known protected area system of Costa Rica, more than 90% of *unprotected* forests are on high or medium productivity lands. Yet only 10% of *protected* forests comprise such lands. This so-called "rock and ice," or "high and far," phenomenon has been documented globally (Millennium Ecosystem Assessment, 2005; Joppa and Pfaff, 2009).

Despite the long history of protected areas and the vast conservation planning literature dedicated to determining where to put them, the empirical evidence base on their effectiveness in reducing pressures is scant and not particularly credible. Much of the empirical evidence falls into two categories: (1) measures of indicator trends (e.g., forest cover) inside protected areas; and (2) cross-sectional comparisons of indicators inside and outside of protected areas. When no ecosystems would be destroyed in the absence of protection, when there is displacement, or when protection is assigned conditional on baseline ecosystem and community characteristics that also affect human use of the ecosystems, such approaches tend to make protected areas look more effective than they are. For example, Andam *et al*. (2008) estimates of avoided deforestation from Costa Rica's protected area network range from 20-50% of the forest protected in an analysis that fails to control for baseline characteristics that affect both deforestation and where protected areas were established. The estimated range falls to 8-12% after controlling for these characteristics.

Fewer than a dozen studies of terrestrial and marine protected areas in developing and developed nations make credible efforts to control for confounding characteristics that affect both the environmental outcome indicators and where protection is assigned. They generally conclude that protection reduces ecosystem disturbance, but at much lower levels than conservation scientists might claim because protection tends to be assigned to ecosystems at below-average risk of disturbance. Only one study considers the potential displacement as a result of protection (as well as potentially positive enforcement spillovers; Andam et al., 2008) and finds no evidence of such displacement. Little empirical research has been conducted on the effects of heterogeneous land-use restrictions across a protected area system (e.g., Sims 2010; Pfaff *et al.*, 2010) or of different protected area manager types, such as comparisons of protected areas run by government, non-profit organizations,

and indigenous communities (Somanathan *et al*., 2009). No study has examined the cost-effectiveness of protected areas compared to other conservation strategies.

Likewise, most studies estimating the socioeconomic impact of protected areas are simple case study narratives or ex ante projections based on extrapolations of historical economic activity, which merely confirm that protected areas are established near poor people and restrict access. Fewer than a half dozen studies observed socioeconomic status and control for confounding factors. Most find no adverse impacts, on average, largely because protection is assigned to unproductive areas (for references, see Andam *et al*. 2010 and Sims, 2010).

The other most common regulatory approaches include land-use restrictions and zoning laws, such as Brazil's laws that dictate how much on-farm forest can be deforested, and species-focused protection laws, such as the United States Endangered Species Act (ESA) or the Convention on International Trade in Endangered Species (CITES). As in the case of protected areas, the theory underlying these regulatory approaches is quite simple, but lack of compliance, pre-emptive destruction, spatial displacement and poor administrative targeting can reduce their effectiveness. Likewise, the evidence base for these regulatory approaches is weak. In one study that controls for some aspects of the non-random assignment of protection, Ferraro *et al.* (2007) find no effect of the US ESA on species recovery in the absence of substantial recovery funding. In fact, their results imply that in the absence of such funding, legal protection leads to adverse consequences for species. One possible explanation is that the ESA encourages private landowners to undertake pre-emptive actions to eliminate listed species, and thus regulations, on their land (the so-called 'shoot, shovel and shut up' response—see Lueck and Margolis 2003). Species-specific funding overcomes these perverse incentives by creating a sufficient level of perceived monitoring and thus credible enforcement.

Regulatory approaches have increasingly been combined with economic incentives to harness the advantages of each approach—building a bridge between command and control and economic approaches to conservation. For example, developed nations have experimented with Tradable Development Rights (TDR), which combine a regulatory cap on the amount of habitat that can be converted with tradable development permits to encourage land-users to meet the regulatory target at least cost. In other nations, protected areas are leased from local communities (e.g., Richtersveld National Park in South Africa; National Park of American Somoa). In Costa Rica, a legal ban on deforestation exists side-by-side with forest protection incentive payments. With the exception of a few TDR programs in developed countries that are focused on open space or farmland (rather than habitat) and Costa Rica's payment system (see review in Pattanayak *et al*. 2010), little evidence exists about the performance of these hybrid regulatory-incentive approaches.

4. Payments for environmental services

What form might a pure economic incentive approach to conservation take? In recent years policy makers and academics have embraced the payments for environmental services (PES) approach to conservation. For example, Dickman et al. (2011) study PECS (payments to encourage co-existence, with, for example, big predators). A key point is that one needs a blend of mechanisms, and that amongst these payments for conservation deliverables are increasingly looking promising.

Wunder (2005) defined a payment for an ecosystem service as (i) a *voluntary* transaction where (ii) a *well-defined* ecosystem service (ES) is (iii) being bought by a (minimum one) ES buyer from (iv) a (minimum one) ES supplier, (v) if and only if the provider secures ES provision. The transaction should be voluntary and the payment should be conditional on the service being delivered – this is no handout to the poor and needy, but

payment for a genuine service. Note, however, that paying for an ecosystem service is not necessarily the same as trading nature on a market. Markets may play a role, as will become clear, but because many of the ecosystem services come in the form of a public good, we cannot rely on markets alone. Governments and intergovernmental organizations are essential to the package.

It is useful to work towards some form of PES classification system. We can distinguish between cases where the ecosystem service benefits a small group of agents versus cases where it benefits a large and presumably more diverse group of agents. If the number of people increases, for example because we are considering regulatory services that impact everybody, the ecosystem services more closely starts to resemble a club good or a public good. Another useful distinction is between cases where service "suppliers" and "demanders" are geographically located close together, and those cases where they are not. Table 1 provides a few examples, highlighting the broad variety of possible cases.

Table 1: Different PES cases

	Local service linkage	*Cross-border service linkage*
Few beneficiaries	*CASE I*: Water and tourism companies	*CASE III*: International users of watershed services
Many beneficiaries	*CASE II*: Urban drinking water users	*CASE IV*: Biodiversity conservation and climate-change mitigation

Case I (local, few users) occurs, for example, when a micro-watershed holds a single main corporative user (e.g., a hydroelectric company or brewery), or when an ecotourism company pays a local community for not hunting in a forest used for wildlife viewing, as happened in the Cuyabeno Wildlife Reserve in Ecuador (Wunder 2000). Many such examples exist,

especially in watershed management (e.g., Porras *et al.* 2008; Wunder *et al.* 2008). Case II (local, many users) is common in macro-watersheds and/or cases with multiple users, e.g, drinking water users in a mega-city combined with irrigating farmers and water-using companies. The challenge of aggregating user interests into payment flows can be met by user associations (the case of 'club goods'), by water utilities (forcing individual consumers to pay), or the public sector (municipalities). Municipal watershed schemes have been mushrooming particularly in Latin America (Southgate and Wunder 2009). Another well-cited example is PES in the Catskills watershed, from which New York City gets most of its drinking water (Ashendorff *et al*. 1997).

Case III (few users of a cross-border ranging environmental service) is the rarest category. There are some ongoing efforts to generate PES systems, for example for conservation of the Danube river system and its biodiversity, or avoiding the siltation of the Panama Canal, which would hurt the international shipping industry. However, in most cases international payments also link to global-level environmental services (Case IV), such as carbon trading around the climate-change mitigation services, or biodiversity conservation initiatives. Just like in Case II, PES schemes will only emerge here if the multiple beneficiaries find means and ways to aggregate their interests into a functional service-buying organ. This can be the case of private companies (e.g. the Dutch FACE Foundation buying carbon credits), multilateral organizations (e.g. the Global Environmental Facility paying for carbon- and biodiversity-enhancing actions), non-governmental organizations (e.g. a nature conservation organization running a biodiversity protection PES) or national governments (e.g. Costa Rica's national PES paying for biodiversity and carbon benefits).

As we can see, the universe of existing and emerging PES initiatives is defined by both the range of the underlying externality (local vs. global) and the number of beneficiaries (few vs. many). Empirically, we see among real-world PES a clear dominance of local-range

club services (watersheds, recreation) and global-range, quasi-public services (biodiversity, carbon). Since most PES schemes emerge from the buyer side, aggregating multiple service users into functional service buyers thus becomes a main challenge for PES developers. For instance, nature conservation organizations like The Nature Conservancy (TNC) and Conservation International (CI) have in Latin America invested in starting up water funds and payment schemes designed to jointly conserve watersheds that are strategic for both water and biodiversity purposes (for examples, see Wunder and Wertz-Kanounnikoff 2009).

Why has PES lately received so much attention? Basically, decision-makers worldwide have increasingly realized that politically convenient win-win projects, allegedly benefiting both local land users and the environment, are hard to implement in practice. Environmental degradation tends to have a local economic rationale, and counteracting it is hence associated with local losers. PES are a tool for natural resource management that explicitly recognizes these contradictions, and uses compensation to bridge between losers and winners.[5] Secondly, by focusing on a specific utility to beneficiaries, rather than nature in its broader and abstract form, PES are also believed to have the potential of eventually becoming more efficient than more indirect conservation tools. Thirdly, compensating the losers is also an equitable way of doing conservation, so PES should have a greater probability of being adopted than command-and-control approaches that directly inflict costs on environmentally degrading stakeholders. Finally, compared to the bipolar character (legal vs. illegal) of most command-and-control measures, or the predetermined investments related to Integrated Conservation and Development Projects (ICDP), PES are a more flexible conservation tool, since the agreed-upon compensations can be renegotiated in response to the rapidly changing benefits and costs in a dynamic world.

[5] Economists often refer to the Coase Theorem in this context: PES try to facilitate exchanges that internalize externalities where property rights are clearly defined and where transactions costs are relatively low.

5. Economic incentives as the road forward?

The discussion until now suggests economic incentives are a valuable instrument in the toolkit of policy makers interested in promoting sustainable resource management and nature conservation. However, this chapter would not be complete without a short discussion of several challenges that should be considered. Some of these may, however, apply to virtually all forms of environmental regulation. For example, often we don't understand the 'production function' of ecosystem services. Consider an effort to re- or afforest agricultural land in order to obtain watershed benefits. How does forest conservation contribute to various hydrological benefits? This is a topic clouded by myths. In the absence of sufficient ecological understanding, both command and control and incentive-based regulatory efforts may be doomed to fail.

Conservation efforts may also be undermined by slippage or leakage. This captures the idea that "successful" conservation schemes may lead to their own demise because they trigger behavioural changes. Consider the effect of a policy where African landowners are paid to remove fences (enabling seasonal migration of wildlife). If such transfers enable landowners to purchase or hold more livestock, then competition for food between wildlife and cows or goats intensifies and the conservation gains from new habitat are eroded (Bulte et al. 2008). Similarly, a law prohibiting fuelwood uses from natural forests could increase pressures on plantation forests, which is another example of leakage.

Occasionally leakage may occur via the market or other mechanisms. If farmers are encouraged to convert farmland back into nature, then local food prices may increase if local markets are not integrated into regional or national ones, inducing other farmers to convert new areas of habitat into agricultural fields. So, gains of habitat in one place may be (partially) offset by losses elsewhere (e.g. Wunder 2008).

In addition to these generic challenges, each type of intervention has its own strengths and weaknesses. Since PES is gaining momentum as the instrument of the future, it is worthwhile to pay special attention to some specific challenges relevant for this tool. First, farmers (and plots) are heterogeneous, and information about local conditions or preferences is privately held. That is, farmers have an information advantage over the regulator that they can use to their advantage. They are better informed about the quality of their land, and will strategically try to retire their least valuable lands, or the ones that they would have retired even in the absence of the payment ("zero additionality"). Generally, this asymmetric information will give rise to so-called information rents, and these will accrue to the better informed party—the farmers (see Ferraro 2008). The cost per unit of nature conserved therefore goes up. Basically, the asymmetry problem occurs as a mirror of the voluntariness of PES. Because farmers themselves have to decide to sign up, there may be an adverse selection problem that reduces additionality. These problems can be overcome by increasing environmental service buyers' information and by careful spatial targeting of PES, often combined with differentiated payment rates per land unit.[6]

There can also be a problem related to the voluntariness of payments on the buyer side. It is not always obvious who will pay for the ecosystem services. This is less of a problem for the cases with "few demanders" who can agree among each other, but negotiations may fail in the presence of public goods with many users, where transaction costs are high and free-riding incentives prevail. When considering about services within a nation's borders, it is thus obvious that governments can play an important coordinating role. Using tax dollars to pay for public goods can in those cases be a solution, so that the government

[6] Additional issues emerge when, from a nature conservation perspective, we care about securing congruent plots of land. How can we convince neighboring farmers to retire adjacent plots of land for the PES scheme? Presumably this will also involve additional costs (in the form of an agglomeration bonus). For example, aggregated farmers deliver better moth biodiversity under agri-environment schemes—see the chapter on butterflies and moths in this Volume.

purchases the ecosystem service from the supplier on behalf of society at large. An example are agri-environment schemes in many European countries. When we move to the global sphere, this role could be played by international organizations, but we currently lack good institutional mechanisms to broker deals between suppliers of ecosystem services and the rest of the world.

PES require that land or resource managers can be identified, who through the provided incentive will change behaviour, compared to business as usual, and thus provide additional environmental services. In developing countries, however, rules of tenure and access to land and natural resources resource may not be clearly defined – especially in agricultural frontier areas where weak governance and environmentally sensitive land areas overlap. It may thus not be possible to find out whom to pay, in ways that are legitimate and efficient, in which case PES may not be feasible.

Finally, it is important to remember the difficulty of simultaneously alleviating poverty and promoting conservation, per the Tinbergen principle mentioned earlier. While PES may be best suited to achieve such win-win outcomes, compared to other regulatory approaches, unfortunately the link is not automatic.[7] Some 1.2 billion people are living on less than a dollar a day (Chen and Ravallion 2007), and many of these poor are found in rural areas, especially in marginal areas like steep slopes of the upper watershed that should be tackled from a PES perspective. Indigenous groups and impoverished rural communities sometimes own or manage sizable shares of tropical forests. The distribution of gains is obviously directly related to the *ex ante* distribution of control over resources generating the services for which compensation is paid.

[7] Note that some of the effects discussed above in the context of leakage are also relevant when analyzing the impact on poverty. The poverty effects are generally ambiguous, depending on whether labor demand goes up or down (affecting local wages), and whether or not the price of food crops changes (affecting the purchasing power of households depending on such crops).

Regarding PES and poverty we can distinguish between two perspectives: (i) the micro perspective of payments and the fate of poor households (participating in the scheme or otherwise), and (ii) the macro perspective of nature conservation as part of an economically viable development trajectory for the economy as a whole. Not much work has been done to address the issue of PES and the national economy (presumably because PES efforts are currently too small to have significant aggregate impacts). As just one example, Norton-Griffiths and Southey (1995) estimated the opportunity cost of biodiversity conservation to the Kenyan economy at 2.8 percent of national income. However, we may expect that nature protection produces little in terms of spillover benefits from which other sectors in the economy benefit. Matsuyama (1992) developed a model of economic growth of a two-sector economy, the manufacturing sector with increasing returns to scale at the sector level, and the agricultural sector subject to constant returns to scale. If we consider 'production of ecosystem services' on a par with agriculture, PES could have the adverse and unintended effect of 'locking' economies into economically suboptimal development paths.[8]

The micro perspective gives rise to somewhat more optimism, but here too we should be cautious not to expect too much (e.g., Pagiola *et al*. 2005; Wunder 2008a). A pre-condition for beneficial effects is that the PES program actually reaches the poor: they should (i) be in the "right place," (ii) want to participate (e.g. it should 'fit' into the poor's prevailing livelihoods), and (iii) be able to participate (e.g. they should be able to make the necessary investments, have sufficiently secure tenure, etc.). It is quite easy to design programs that fail on either of these conditions, so that PES does not help in fighting poverty. Next, *if* PES reaches the poor we may infer that it will make them better off. Participation in a PES scheme is voluntary so participants should gain, at least in expectation. The proper net measure of benefits for the eco-service seller is payment received minus opportunity cost.

[8] But note that, in other cases, where PES pays for the restoration of environmental services (e.g., on degraded lands), it may also have positive multiplier effects, turning this argument on its head.

When payments are aligned with opportunity cost, the net benefits may be small (although the stabilizing effect of payments on household income is welcome). But additional effects may exist. In the case of imperfect tenure security, PES schemes could potentially induce powerful stakeholders to "muscle out" poor households; conversely PES could help poor people to consolidate tenure *vis-a-vis* external introducers, and stimulating better community organization. Empirically, the latter, positive impacts have been dominating (Pagiola *et al*. 2005, Wunder 2008a).

In the light of these challenges it is perhaps no surprise that the empirical evidence on the efficacy of PES is rather ambiguous.

6. Application: REDD

Climate-change mitigation is high on the international agenda, and so is the most favoured forestry response—Reduced Emissions from Deforestation and forest Degradation in developing countries (REDD). In the latest multilateral negotiations around the Kyoto Protocol, the concept has been widened to REDD+, where forest regeneration and enhancement of the carbon stocks of standing forests are also being integrated. The Stern Report (Stern 2006), and subsequent work done in particular by the consultancy firm McKinsey (see http://www.redd-monitor.org/tag/mckinsey/ for a critical assessment) have pointed to REDD+ projects as some of the cheapest mitigation actions that could be taken in the short run: buying out extensive converted uses of forestlands with low profitability is allegedly much cheaper than many of the mitigation options in other carbon-emitting sectors. Hence, with the UN, the Word Bank, and other agencies as facilitators, many tropical countries are currently getting ready for receiving REDD+ assistance on a supposedly large scale.

Conceptually, we can think of REDD+ as a kind of international PES scheme, with conditionality as a key design feature (Angelsen *et al.* 2009). The idea is that forested countries receive periodic economic compensations for deforesting less than they otherwise would have done. The size of REDD+ payments will depend on the achievements the country has made in stabilizing its forest cover, compared to a (yet-to-be-defined) baseline counterfactual. Some incipient bilateral mechanisms of this type are already in place, such as the agreements Norway has struck with Brazil, Tanzania, and Indonesia. Only time will tell how seriously the *quid pro quo* design will be taken by donors and recipients. As with government-coordinated PES programs, the political costs for donors of withholding money are often very high, and therefore some non-compliance is likely to be ignored.

How would tropical REDD+ recipient countries then make sure that deforestation is actually being curbed, so as to qualify for a continuous flow of conditional REDD+ funding? Certainly, PES agreements with landowners on the ground could be one possibility for REDD+ recipient countries to directly pass on carbon credits (and land-use obligations) to the sub-national level. However, in those cases where forestland tenure is ill-defined, PES may not be the right solution. One option will be facilitating investments that better enhance forest conservation, such as land reforms and improved governance. Yet, disincentive measures, such as increased forest law enforcement, or creating new protected areas (and better manage existing ones) are also options to spend REDD+ money in ways that can actually curb forest clearing. Finally, REDD+ resources also make it possible to achieve leverage at the policy level, for instance compensating municipalities for not receiving a new road into the forest, or incentivizing the cattle farmer association to cooperate on REDD+ with subsidized credits for pasture intensification. The international debate has so far focused more on benefit sharing questions, widely by-passing the fact that deforestation will not just stop by itself, making targeted investments necessary to make REDD+ work.

7. Conclusions

The conversion of natural capital into alternative forms of capital continues, threatening increasingly the supply of valuable environmental services. While part of this conversion process is economically rational—alternative forms of capital simply yield greater (discounted) returns—undoubtedly some destruction is socially wasteful, and may be accelerated by market or policy failures. From both an intergenerational welfare and an ethical perspective (including non-anthropocentric concerns), policy interventions to slow down or reverse natural degradation are warranted.

But which policy instruments should be used for this purpose? Traditionally, command and control style regulation has been dominant. In the domain of biodiversity conservation, regulation has mostly relied on protected areas, banning certain forms of land and resource use altogether. However, mounting pressures—population growth and economic expansion—have spurred increasingly active searches for alternative, supplementary approaches to conservation. Economic instruments are one such supplement – and in general a promising one, we believe. Standard economic instruments, such as taxes, subsidies and tradable permits, have been applied (with varying success) in the domains of pollution control or the regulation of resource extraction. One specific economic instrument, payments for environmental services, has recently entered the domain of biodiversity conservation.

PES have many advantages, and their voluntary natures make them more socially palatable. They can be applied at the local level between private parties, for example when local consumers or an electricity company compensate uphill farmers for providing valuable watershed services. The same principles can also be applied at the global level, even if this typically involves public parties as well as private ones. For example, via REDD+ schemes

PES may come to play a significant role in global efforts to mitigate the adverse effects of the greenhouse effect.

Obviously, economic instruments are not a panacea. Leakage may partially erode conservation gains, information asymmetries between private parties and regulators may introduce information rents (raising the costs per unit of nature conserved), and institutional weaknesses—especially at the international level—may limit the scope for large scale and successful application. However, the flexibility of economic instruments and the implied efficiency benefits suggest they will have an increasingly large role to play in the portfolio of conservation activities. Moreover, many of the potential shortcomings of economic instruments can be overcome through specific targeting and more fine-grained regulation, as we learn to improve the design of these instruments. Yet for this purpose, we will also need rigorous impact assessments and modelling to better understand "what works when, and for whom?"

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