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Conservation of Tropical Forests: Addressing Market Failure

Erwin Bulte and Stefanie Engel

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1. Background and motivation

Global forest stocks have fallen only very slightly in recent years. However, behind this re-assuring aggregate statistic one can detect conflicting trends in tropical and non-tropical regions, and opposing tendencies in natural forests and plantations. While plantation forests in the developed world are increasing, natural forests in the tropics continue to disappear at a rapid rate. Using the FAO definition of deforestation,[1] the average annual deforestation rate over the period 1990-2000 was 0.2% for the world as a whole, but 0.4% for South America, 0.8% for Africa, 1.2% for Indonesia and 1.4% for Myanmar (FAO 2001). Upon converting rates to hectares, annual net tropical deforestation proceeded at an average speed of some 11.8 million hectares for the period 1981 to 1995. Forest loss rates vary greatly across different countries (WRI 2000), but extensive deforestation has taken place in Asia (e.g. Indonesia, the Philippines, Thailand), Africa (Ivory Coast, Cameroon) and Latin America (Colombia).

Despite these large accumulated losses, considerable areas of tropical forest remain. In 1995, tropical forests were estimated to cover an area of about 1,734 million ha or about 13.4% of the globe's land area, excluding Antarctica and Greenland. With sizable tropical forest areas remaining, and with forest stocks expanding in other regions of the world, why should tropical deforestation be considered a problem at all?

The gradual disappearance of such forests, whether matched by expansion in plantation areas elsewhere or not, should be alarming to society for at least three reasons. First, tropical forests contain much of the world's biodiversity. Tropical forests, especially wet tropical forests, typically contain far more species than their temperate counterparts, and they are thought to account for perhaps two–thirds of the earth's approximately 14 million species (Hughes *et al.* 1997). Second, tropical forests represent a considerable store of sequestered carbon which, when released, would contribute to climate change. Third, tropical forests are important for forest dwellers and many rural poor alike, who depend on access to forest services for their survival.

In addition to these benefits, tropical forests generate various other tangible and non-tangible outputs, to which we will return in detail below. It is obvious, however, that forest conservation and sustainable natural forest management not only imply benefits – they may also invoke considerable costs. Costs are mainly foregone returns to alternative forms of land

[1] The FAO defines tropical deforestation as occurring when canopy cover is reduced to 10% or less. Note that not all deforestation is necessarily permanent. For example, when cleared lands are abandoned after a period of cropping, secondary forests may develop (to a greater or lesser extent able to 'substitute' for primary forests).

use, but may also include nuisance costs of wildlife invading adjacent agricultural fields or negative hydrological impacts by competing for water with other land uses.

A well-known but important observation concerns the spatial and temporal mismatch of costs and benefits. Costs occur often at a local scale and must be borne in the short term. Households, firms, and governments may be confronted with a variety of current costs and restrictions, in some cases impeding the ability to generate income of those who may need extra income most. In contrast, benefits might not materialize until far in the future, and could be shared by many parties, possibly even at a global scale. Globally shared benefits are, for example, the contribution of forests to global climate regulation, or their role in protection of charismatic wildlife. When uncompensated, such benefits might constitute a windfall gain for the rich.

Market imperfections and policy failure frequently prevent an efficient forest management at the national level. Moreover, even if sovereign nations have the institutional capacity in place to allocate their land optimally to various uses, the spatial and temporal mismatch described above implies the following. Unless either a full set of markets exists for all forest ecosystem services or a co-operative outcome can be negotiated, in the long run, sovereign nations will ignore spillover benefits and as a consequence will allocate too much of their land to uses other than forest. While rational from a narrow private or domestic perspective, in light of the inability to capture a reward for spillover or external benefits, this means an efficiency loss from a global perspective. In addition, when the institutional capacity to steer land use is lacking, the mismatch implies that sovereign countries have less incentives to establish such an infrastructure. Market failure may therefore result in institutional failure. One possible outcome is that property rights to land and forest resources are *de facto* undefined (or un-enforced), such that individuals will ignore the user cost associated with current extraction or conversion, and refrain from investments in future productivity of the resource. This can result in uncontrolled forest conversion for agricultural uses (as described, for example, by Myers 1994), but also in large-scale illegal logging for export markets. For example, Mintorahardjo and Setiono (2003) report that in Indonesia 60 to 80 percent of the timber industry's supplies are obtained illegally.

Recognizing this threat, the international community has developed various instruments to align national and global incentives. Instruments vary from the carrot to the stick, ranging from transfers and debt-for-nature swaps to timber trade restrictions. The international community also does many things that change the incentive structures indirectly by affecting international prices and global investment flows. Ongoing tropical deforestation suggests that these efforts have only been partially successful at best. Upon closer inspection of the available evidence (to which we return below), it is tempting to conclude that 'the North' wishes to free ride on conservation efforts in 'the South,' and fails to develop the appropriate institutions to internalize transboundary and intertemporal external effects associated with current forest conservation. Recent initiatives of direct payments for environmental services appear to be a promising step in the right direction. However, we argue that the implementation of these and other policies needs to take into account the property right regime in place in the country of concern. Property right regimes have changed considerably over the last decades, with strong trends to devolve rights and responsibilities over local forest resources from the state to communities and private individuals. This has important impacts on the relative importance of different types of market failures and policy effects.

This paper proposes to (1) summarize existing literature on the causes of deforestation and the magnitude of the various forest benefit components (enabling an assessment of the importance of spillover effects in an absolute and relative sense); (2) critically discuss the usefulness of forest valuation exercises for guiding policy choices regarding forest management and conservation; (3) discuss the main market failures underlying deforestation and the policy approaches used to address them; and (4) highlight the relationship between different property rights regimes, market failures, and policy effectiveness.

The paper is organized as follows. In section 2 we briefly summarize the existing results on the immediate and underlying causes of deforestation. From our discussion so far it is clear that deforestation *per se* is not necessarily inefficient. The socially optimal level of deforestation should equate the social marginal benefits of deforestation to the marginal costs. Therefore, in section 3, we review the various forest functions and services that have been identified in the literature, and critically discuss the potential of forest valuation work as a tool to inform policy makers in the real world. In section 4, we discuss the market failures that are the root cause of many deforestation problems, and various specific policies aimed to address them. As argued above, different property right regimes are affected differently by market imperfections, and thus, effects of policies vary across these regimes. These relationships are discussed in section 5. Finally, section 6 concludes.

2. Causes of deforestation

In an extensive review of existing economic models of deforestation, Kaimowitz and Angelsen (1998, ch. 2) distinguish between sources of deforestation (i.e., the agents and actions causing deforestation), the immediate causes of deforestation (i.e., the decision parameters and agent characteristics driving deforestation decisions), and the underlying causes of deforestation (i.e., macro-level variables and policy instruments affecting decision parameters).

Main sources of deforestation include the expansion of cropped area and pasture, logging (particularly in Southeast Asia) and to a lesser degree fuelwood collection (particularly in Africa). The main agents of deforestation also differ across regions. In somewhat simplifying terms one can say that they are smallholders in Africa, timber companies and plantation agriculture in Southeast Asia, and cattle ranchers and mechanised farmers in Latin America. Evidence on the effect of agents' time preferences, risk aversion and wealth status on deforestation is generally weak and contradictory (ibid, ch. 6).

The parameters driving deforestation decisions include physical characteristics of the forest, agricultural input and output prices, timber prices, wages and off-farm employment, technological change in agriculture, accessibility of the forest, as well as the property regime and strategic behavior. Deforestation tends to be stronger under ecological and economic conditions more suitable to agriculture (e.g., higher fertility, higher agricultural output prices), although changes in relative prices that affect crop composition may have ambiguous effects. The effect of timber prices depends crucially on tenure security. If producers have insecure rights to the forest, higher timber prices are likely to lead to more deforestation because they increase the net benefits from deforestation. With secure tenure, however, higher timber prices may increase the incentive to adopt more efficient harvesting and processing techniques, and to guard against encroachment. Kaimowitz and Angelsen (ibid) argue that under most developing country conditions, logging and agriculture have to be seen as complementary rather than competing activities (what they call the 'logging-shifting cultivation tandem') and that higher timber prices will likely increase deforestation. While these authors conclude that increases in rural wages and off-farm employment opportunities tend to reduce deforestation, the opposite may be the case in situations where labor is abundant relative to land (López, 1998) and where communities have to compete with firms for property rights (Engel and López, 2004). Increases in farm implement prices appear to decrease deforestation, but the effect is less clear for fertilizers (Kaimowitz and Angelsen, 1998). Technological change generally has an ambiguous effect on deforestation, with technologies that are particularly suited for land already under cultivation rather than land under forest cover and that are intensive in both labor and capital most favorable to reduce deforestation (ibid, López, 1998, Angelsen and Kaimowitz, 2001). Improved access to markets and forests (e.g., through roads) generally increases deforestation. Well-defined and secure property rights reduce deforestation as compared to open access situations. Where farmers obtain property rights by clearing land, improvements in tenure security, however, may increase deforestation (Kaimowitz and Angelsen, 1998).

Potential underlying factors of deforestation include population growth, economic growth, external debt, trade, and structural adjustment, but empirical and theoretical evidence on the direction of the effects is ambiguous and depends on the specific context. Political economy considerations are another type of underlying factor that have received relatively little attention in past research (see also the chapter by Deacon and Mueller in this volume). Often, incentives are distorted in favor of excessive forest exploitation in order to serve vested 'resource mining' interests (Wunder, 2000). The political scientist Ross (2001) has described how, following a boom in timber prices, incumbent politicians may deliberately demolish the institutions that promote sustainable logging. The objective of such politicians is to maximize their own private profits from logging, or the value of bribes associated with allocating timber licenses to favored parties – a process Ross refers to as "rent seizing."

Again, it is important to note that deforestation *per se* is not necessarily inefficient. The socially optimal level of deforestation should equate the social marginal benefits of deforestation to the marginal costs. However, evaluating these, especially the benefits from forest services, is not an easy task. We now turn to an assessment of these benefits and the potential for economic valuation to assist policy makers in this regard.

3. Forest Services and Valuation

It is probably fair to say that without the plethora of services provided by natural ecosystems, mankind would not be able to prosper the way it does. In fact, it is conceivable that it would not survive at all. However, this observation does not imply that every natural ecosystem should be kept intact regardless of the costs. While ecosystem services are presumably infinitely valuable to human society, it may be equally true that conservation of natural ecosystems is a bad investment from societies' point of view. This apparent paradox is readily resolved. Generic statements about the value of ecosystems are an assessment of the *total value* of all ecosystems – how much will mankind lose when all ecosystems are converted or degraded? Of course this is an unlikely scenario in the short run, and one that is therefore essentially irrelevant for policy purposes. Investment opportunities are typically evaluated at the *margin* – what is the value of one more hectare of boreal or tropical forest, given that there exists a pre-existing stock of ecosystems? If the pre-existing forest stock is large (and the total value of forest services enormous), the value of an additional stretch of forestland may well be negligible. What services do forests provide, at the margin and otherwise?

3.1 Ecosystem services

Many biophysical and biochemical processes take place within natural ecosystems. The outcomes of some of these processes are useful or beneficial for mankind. In what follows we define such outcomes as *ecosystem services* (ES). There are various ways to cluster or organize the various ES, and any clustering is arbitrary. In this section we distinguish between production services of tropical forests, regulatory services, and habitat/biodiversity conservation services. Drawing from earlier work by van Kooten *et al.* (2000) and Nasi *et al.* (2002), we discuss each of these classes in turn below.

Before starting off, however, a word of caution is in order. It proved impossible to provide sensible monetary estimates for the various ES at the current level of aggregation – the set of tropical forests is simply too diverse. For example, consider the forest service of timber production, one of the key services generating cash income and one for which information is readily available. Rents from harvesting may vary from virtually nothing to some $1,000 per m^3, depending on the location and harvest techniques employed. Similarly, the quantity of timber that can be sustainably extracted from natural forests ranges from some 0.5-2.0 m^3 ha^{-1} (in countries like Costa Rica) to some 20 to 30 times that quantity (in Indonesian *dipterocarp* forests). Given this broad range of outcomes, presenting monetary estimates for tropical forest management 'in general' serves little purpose. A particular case study, for which a consistent set of data is available, will be provided in section 3.

Production Services of Tropical Forests: Tropical rainforests produce tangible products such as timber, fuelwood and non–timber forest products (e.g., rattan, oils, fruits, nuts, ornamental flowers, bush meat), but also less tangible assets such as opportunities for eco–tourism. Timber production is commercially the most significant activity in most forests. There exist huge differences between very selective logging of high value hardwoods, more intensive logging of non-dipterocarp forests, intensive logging of dipterocarp forests, clear felling in natural tropical softwood forests, and clear felling of mixed tropical hard woods for pulp production. The returns per hectare vary accordingly – they not only depend on the type of forest and management regime, but also on discount rates, stumpage prices, management costs, site conditions, distance to markets, infrastructure, etc. Vincent (1990) provides estimates of present value ranging from +US\$850 down to –\$130 ha^{-1}. Selective logging may be a commercially attractive proposition in some areas, but not elsewhere. For example, recent experiences in countries like Bolivia suggest that annual profits from selective logging are smaller than \$1 ha^{-1} yr^{-1}; logging firms simply opted to return their concessions when confronted with such taxes (Bojanic and Bulte 2002).

Fuelwood rarely enters international markets, but is increasingly produced for regional and national markets. It is one of the most important forest inputs in poor households in developing countries. The mid 1970s were marked by a widespread concern that demand for fuelwood and charcoal outpaced supply. This was perceived to result in a "fuelwood gap" with disastrous consequences for rural poor and forests alike, which triggered interventions aimed at expanding supplies (farm forestry, plantations) and lowering demand (enhance fuel efficiency, etc.). According to the FAO (2001), some 1.6 billion m^3 of fuelwood were extracted in the mid 1990s, a number which is now believed to decline slowly.[2] While an estimated 2.4 billion people continue to utilise wood and other forms of biomass, recent assessments indicate that woodfuels are unlikely to be a major cause of deforestation (Arnold *et al.* 2003). Urban consumers substitute away from biomass to alternative sources of energy when they can afford it (hence, actual consumption is smaller than predicted), and much of the supply now comes from non-forest areas.[3] While fuelwood collection is expected to remain a major source of

[2] Charcoal consumption, on the other hand, is still increasing, and now requires some 270 million m^3 of wood per year.
[3] But deforestation can occur locally in the vicinity of growing urban markets with little purchasing power (typically in Africa).

income for millions of rural poor households, harvesting will take place in the vicinity of populated areas and its scope for application over extensive areas in natural forests is limited. Wood is a bulky commodity, and the option to profitably gather it is restricted by harvesting and transport costs.

Another potentially important ES concerns the generation of <u>non–timber forest products</u> (NTFP) such as rattan, oils, fruits, nuts and bush meat. Large numbers of forest dwellers depend critically on them for survival, and large numbers of rural and urban poor supplement their income by seasonally moving to the forest to gather commodities like Brazil nuts (Bojanic 2001). Cavendish (2000) presents evidence for Zimbabwe that suggests some 30-40% of household income may derive from NTFPs. NTFP harvesting from unmanaged natural forests has been likened to a 'safety net' to which the poor can turn in times of adversity, but other authors argue that it usually offers little scope for economic development and could perpetuate poverty (Homma 1994). Regardless, a common finding is that the returns of NTFP gathering on a per hectare basis are very modest. While an early study by Peters *et al.* (1989) proposed returns that could compete with other commercial uses, many authors have since cautioned against extrapolating 'per-hectare' estimates from sample plots to large stretches of tropical rainforests. Downward sloping demand for NTFP, uncertainty concerning sustainable supply, and increasing costs of production and transportation are all limiting factors. For example, the net annual returns to gathering a major international commodity like Brazil nuts in Bolivia amounts to no more than $0.5 per hectare (Bojanic 2001).

Likewise, <u>eco–tourism</u> is only locally important. Although tropical (moist) forests are generally not very attractive to tourists because of the humid climate and their limited scenic value (compared to, say, East African game parks), recreation and tourism have the potential to generate foreign exchange (for an optimistic case study, refer to Ruitenbeek 1989). However, the role of eco–tourism in promotion of forest conservation will likely remain small, and its value will fall on a per hectare basis as more areas are made available for tropical forest recreation.

Regulatory Services of Tropical Forests: Tropical ecosystems provide a wide range of regulatory services. <u>Watershed protection</u> and the provision of <u>hydrological services</u> are clearly important examples in this respect. The surface of denuded forest areas may be compacted by rain – reducing infiltration, increasing runoff and lowering water quality. But exposing areas to the natural elements unprotected is but one possible form of landuse after deforestating an area, and possibly not the optimal one. Types of landuse other than forests can also provide hydrological services, albeit in varying degrees – compare seasonal cropping where the soil is bare during part of the year versus systems with year-round crop cover offering more protection (extensive livestock ranging systems or fruit tree orchards). Ground cover appears to matter more than canopy from an erosion perspective (Wiersum 1984, Calder 1998).

Apart from protection against soil erosion and sedimentation, tropical forests are believed to provide a more balanced supply of water when there are seasonal differences in precipitation because the soil acts as a sponge. However, again, it is important to recognize that tropical forests are not the only ecosystem capable of producing these effects – the nature of the succeeding land use is very important. Equally importantly, as argued by Calder (1998), evapotranspiration rates of forests are high, possibly decreasing water supply from forested areas and lowering water tables. Evapotranspiration rates also have an impact on rainfall patterns, but Calder (1998) argues that it is too simplistic to relate forest conservation to enhanced rainfall. Many other variables will be affected by changes in landuse as well, including surface albedo, air circulation, cloudiness and temperature. For this reason, the magnitude and sign of the regulatory effect is unclear *a priori*, but recent consensus is that this effect is likely of minor importance.

Another regulatory service identified in the literature involves <u>pollination</u> services provided by forest pollinators (mainly insects) and pest control by forest 'predators' preying on potential crop pests. Such pollinators and predators depend on forest habitat for their survival, and their disappearance would impact wild ecosystems and agricultural production systems alike. There exist estimates for the value of natural pollination and natural pest control in the US (both run well in the billions of dollars on an annual basis), but there is little information for developing countries (see Reid 1999). The lack of more information is no surprise, perhaps, as assessing benefits of these services requires insight in potential substitutes for them, which is incomplete at best.[4] Finally, offsetting, and in some regions certainly dominating, the beneficial effects of pollination and pest control are the crop damages caused by forest animals.

A final regulatory service, <u>storage of the greenhouse gas</u> carbon dioxide CO_2, is likely the most important non-market benefit associated with tropical forest conservation. The Intergovernmental Panel on Climate Change (IPCC) provides overviews of estimates of the

[4] There is evidence that some substitution possibilities do exist. For example, when a pollinating bee disappeared from a region in China where apples were grown on large-scale apple orchards, farmers switched from natural pollination to hand pollination. Interestingly, they were unwilling to return to natural pollination when that option was offered later.

social costs of CO_2 emissions in different decades (Houghton *et al*. 1996). It does not endorse any particular range of values for the marginal damages of CO_2 emissions, but cites published estimates of discounted future damage of US$5–$150 per metric ton of carbon emitted, depending on, among other things, the discount rate applied to weight future costs. Assuming a shadow price of US$10–$25 per metric ton, and a release of some 50–140 metric tons after deforestation, the costs amount to $50 to $3,500 per hectare, or $25–$175 ha^{-1} yr^{-1} (using a 5% discount rate). These estimates may even have to be adjusted upward for some areas that hold larger quantities of carbon (up to 250 tons for closed primary forests), and possibly in light of recent arguments that the shadow price of stored carbon may be as high as $34 per ton (Clarkson 2000).

Habitat, Biodiversity and Non–use Values: Tropical forests are home not only to millions of people for whom forests may be an integrated part of economic, social and religious life, but also to millions of animal and plant species, most of which are endemic to the local forest ecosystem. Various species have both use and non–use (preservation) value, and diversity *per se* may also have a role of play in ecosystem functioning and service provision.

- The direct use values of biodiversity have attracted the attention of economists and ecologists alike, not in the least spurred on by the belief that demonstration of high values provides a convincing argument against human intervention in "vulnerable" ecosystems. Thus Leakey and Lewin (1996), for example, describe how lucrative and important the drugs Vincristine and Vinblastine, alkaloids from the rosy periwinkle from Madagascar, have been in curing acute lymphocytic leukemia and Hodgkin's disease. The rainforest may be a valuable source of new medicines, and searching for these uses is usually referred to as <u>biodiversity prospecting</u>.
- Consider pharmaceutical uses. There are some 250,000 species of higher plants, and approximately 125,000 of these are found in tropical regions. To date about 47 major drugs have come from tropical plants. Simpson *et al.* (1996) have investigated the (potential) industrial, agricultural and pharmaceutical values of biodiversity hotspots. Even under the most optimistic assumptions, the value of marginal species is small, less than $10,000 at best. As the number of species increases, the value of marginal species falls—from almost $3,000 when there are 250,000 species to a negligible amount when there are more than one million. If the value of marginal species is small, then, by extension, so is the value of a marginal hectare in biodiversity prospecting. Simpson *et al.* (1996) have computed maximum willingness to pay for biodiversity prospecting for a range of 'hotspots' and their upper bound

estimates range from \$0.2 to no more than \$20 (in Ecuador). Barbier and Aylward (1996) also conclude "the potential economic returns from pharmaceutical prospecting of biodiversity are on their own insufficient justification for the establishment of protected areas in developing countries."[5]

Obviously, limited direct use value does not imply that the economic value of biodiversity is modest. For example, ecosystem stability and resilience may be positively linked to diversity (Perrings 1998). However, empirical research on this topic is still in its infancy, with most empirical work focusing on simple ecosystems (e.g., grasslands) under rather controlled conditions.

In addition to direct and indirect use values, non–use values (including cultural values) are also important. Many people derive utility from the knowledge that tropical forests exist and are home to many species with which we share the earth, even though they will never visit these places themselves or intend to "use" them otherwise. By using stated preference methods such non–use values may be approximated, although this approach continues to be an issue that divides the scientific community. Working with US respondents, Kramer and Mercer (1997) estimated that preservation of global tropical forests had a one–time value of US\$1.9–\$2.8×10^9 (\$21–\$31 per household), or annual value of only \$95–\$140 million (using a 5% discount rate). Conservatively multiplying by four to take into account Canada, Australia, New Zealand and Western Europe yields an estimate of annual existence value of \$380–\$560 million. Assuming constant marginal nonuse values and dividing by the total tropical forest area (about 1,750 million ha in 1990), we arrive at an annual per hectare value of approximately \$0.2-0.32 per ha. Dividing instead by the total area of tropical rainforest (about 720 million ha), existence value per hectare rises to \$0.52-0.78 per year. Since non–use values likely decline at the margin as the forest stock increases, the marginal preservation value is even lower than the average preservation value.

3.2 Assessment of Outcomes and Methods

How would economists use the information about ES to inform policy makers about landuse policies? The standard approach would be to (i) quantify these services and attach a monetary value to them, and (ii) aggregate these values to obtain an estimate of the "total economic value" of sustainable tropical forest management. Forest valuation, then, enables policy makers at least in theory to make informed tradeoffs about the benefits of conservation

[5] In a more recent paper by Rausser and Small (2000) the implications of the work of Simpson *et al* are disputed. By adding efficient search to the theoretical model, Rausser and Little argue that the value of hotspots can be increased enormously. See Costello (2003) for a critical assessment of this claim.

of forests versus those of alternative land use options. We will pursue this approach in Box 1 below.

It is noteworthy to point out that while tropical forests provide a wide range of ecosystem services, the marginal value of forest conservation appears rather modest from the discussion above (something that will be confirmed by the numerical example below). With the important exception of carbon sequestration, the aggregate per hectare value of production, regulatory and habitat services is small. Hence, tropical forests may have a hard time "competing for space" even in the absence of market failure.

While forest services are perhaps essential (infinitely valuable?) for mankind, it should be no surprise that a natural upper bound for the value of forest conservation is defined by the budget constraint. But why are estimated values so low? The predominant reason, presumably, is that forest areas are not yet scarce. Rapid deforestation rates notwithstanding, there are still hundreds of millions of hectares of tropical forest. The nonuse values generated by these hectares are essentially public goods – hectares of nature serve all. The ecotourism opportunities of distinct forest areas are substitutes in consumption, with different forest areas competing for customers. It is expected that as more of the tropical forest is converted to other land uses, the costs of further conversion (the value of foregone ecosystem and other non–timber amenities) will increase as well. At some point, the marginal costs of additional land conversion will equal or exceed marginal benefits, and no further deforestation of tropical forests should occur.

However, there are several caveats and potential problems associated with the economist's approach and, depending on one's politics, these caveats might well pose insurmountable problems. In this section we will present a brief overview of various relevant issues in an effort to qualify some of the numerical results that follow, and emphasize that current low estimates of the value of forest services should be treated with caution. First, note that distributional concerns have been ignored thus far – one dollar of surplus generated by cattle farming in the hand of a large farmer is considered equally valuable as one dollar of surplus earned with rattan collection in the hands of a forest dweller. This is a straightforward application of the Kaldor-Hicks hypothetical compensation principle – if winners of certain activities (such as deforestation) earn more than enough to potentially offset the losers, then that activity is desirable. Economists then assume that distributional concerns can be tackled later, for example by taxing the farmer and compensating the forest dweller. But if the proper institutions for making *actual* compensation transfers are not in place in large parts of the world where tropical forests are found, then it is an open question whether distributional issues can really be ignored when we are interested in welfare comparisons. Since hundreds of millions of very poor people rely on access to forest resources for their survival, and alleviation of their poverty is a widely supported objective, this is a particularly pressing concern.

Second, the discussion thus far has severely downplayed the many uncertainties that surround deforestation. Ecological feedback effects can (but need not) have substantial effects at both the local, regional and perhaps global scale. Some ecological systems have threshold effects such that relatively small causes can trigger large consequences. Moreover, such consequences may to a large extent be irreversible due to hysteresis effects. It remains to be seen whether such concerns are germane to tropical deforestation, and in the interim it might therefore be advisable to err on the cautious side when converting forests. The precautionary principle certainly suggests this, and conventional economic science has few alternative concepts to guide social decision-making when trying to come to terms with unforeseen outcomes of unknown probabilities. One potentially useful concept in this context may be quasi-option value (e.g. Albers *et al*. 1996, Albers 1996).

Third, and related to the issue above, when we account for uncertainty in forest conservation it becomes evident that we are not really interested in how the value of certain forest services may fluctuate over time. Rather, society should care about the *overall* risk of its total investment portfolio, of which natural forests are only one element. How are fluctuations in the value of forest services correlated to fluctuations in other portfolio components? This issue appears essentially unknown and requires more research.

Fourth, there now exists a large body of experimental literature that suggests that people value gains and losses differently, even though economic theory postulates that gains and losses should be valued identical (apart, possibly, from an income effect). Knetsch (1993) notes that people evaluate gains and losses not strictly in terms of end states, but also in terms of changes from some reference situation – say; the current forest stock. Willingness to pay (WTP) for forest conservation may therefore be quite different from willingness to accept (WTA) compensation for ongoing deforestation. If indifference curves are indeed 'kinked' at the endowment level, as suggested by some of the empirical evidence, then the issue of who owns the forest becomes of paramount importance. Are tropical forests owned by sovereign nations, or are they part of a global heritage for which shared ownership and responsibility applies? WTA for deforestation is unbounded by current income, and could well exceed the gains from further forest conversion.

Fifth, and perhaps most importantly, while the problem of valuing regulatory functions is intrinsically related to the diversity of effects and consequences, and the uncertainties that surround them, a major problem in valuing habitat functions of tropical forests is rooted in ethics. The species-area curve may be used to relate deforestation rates to extinction rates, and it is clear that there exist alternative philosophical stances on trading off species versus human welfare. Conventional economics need not take pre-eminent status when deciding about nature (Johansson-Stenman 1998). As argued by Sagoff (1988): "it is not just a matter of balancing interests with interests, it is a matter of balancing interests with morality, and balancing one morality with another morality." Common (1995) also notes that economics is particularly important in developing instruments to achieve certain objectives (such that these objectives are reached cost effectively), but not necessarily to determine the objectives themselves.

In addition to these rather generic concerns about valuation of forest services, there are many other practical issues that emerge when balancing the benefits of sustainable forest management against those of other forms of land use – particularly about time trends of key variables. We will consider this in Box 1 when we present data for the Atlantic zone of Costa Rica. With these caveats in mind, it is left up to the reader to decide whether forest valuation represents a useful exercise or not.

Box 1: Uncompensated Spillover Effects – An Example from Costa Rica

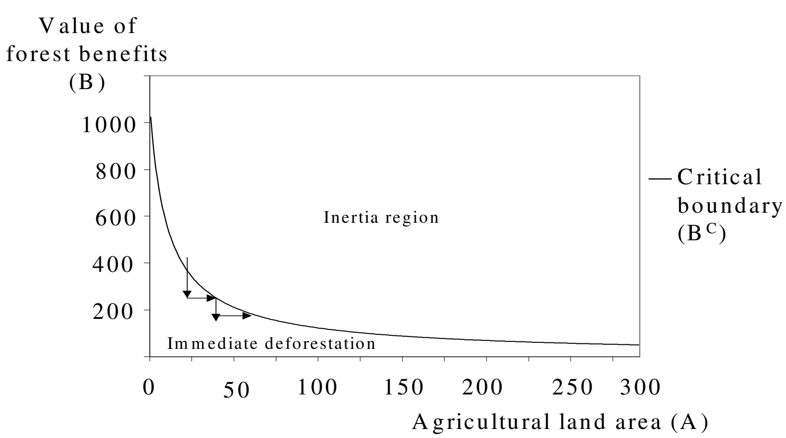
In this section we will numerically illustrate some of the main issues raised above, and compare the total economic value of forest services to the benefits that can be obtained when converting the land to some other use – agriculture in this case. This obviously requires information on forest values, but also on the benefits of agricultural conversion. We briefly deal with both issues in turn, using data from the Atlantic zone of Costa Rica (a study region of some 320,000 ha, for details and caveats refer to Bulte *et al.* 2002).

Benefits of agricultural expansion

The benefits of agricultural expansion (or the opportunity cost of forest conservation) are approximated by a large LP land management model, discussed at length in Schipper *et al.* (2000). The model combines detailed agronomic information and economic data (demand for commodities, supply of factors, transport cost, etc.). The total land base L can be allocated to either agricultural land A or remain as forest area F. The focus is on maximizing the present value of net social benefits, or the sum of consumer and producer surplus. We assume a benevolent dictator exists that can freely allocate land to its most productive use – the use that maximizes welfare. It is assumed that those areas most profitable for agriculture are taken in production first. Then, as the agricultural area expands, increasingly less attractive areas are encroached upon. Upon applying a step size of 1,000 ha, the LP model can be used to simulate land use and compute its returns on a per-hectare basis as the agricultural frontier moves out. This simulated output is then used as an input in a regression analysis. We explain the "shadow price" of land in agriculture P^A as a function of the agricultural land base A. This yields as the preferred specification:

(1) $\ln(P^A) = 15.8 - 0.89\ln(A)$,

where both coefficients are significant at the 1% level. Plotting equation (1) in a graph illustrates how the value of agricultural land changes as deforestation proceeds.



The line in this figure is the estimated shadow price of agricultural land, which is a decreasing function of the land base in production. While the benefits of cropping in prime regions are very high (think of growing bananas or palm heart on preferred soils), it is evident that returns to agricultural expansion drop sharply after the most suitable soils have been taken in production. The per hectare returns to extensively managed natural pasture systems, the flat end of the shadow price curve, are low.

Box 1 continued

Benefits of forest conservation

We assume the study region in question to be sufficiently small for the marginal value of forest conservation to be constant. We use best available evidence published in the literature. Typically these data are based on average values of the ES discussed above – preferably for the study area but if such information was unavailable we used evidence for Costa Rica or, if that was lacking, for Central America. Bulte et al. equate forest conservation with sustainable forest management (i.e., they allow some timber and NTFP extraction and ecotourism), and as a baseline approximation they assume that the total economic value of forest management amounts to \$200 ha^{-1} yr^{-1}. This number can be broken down in its elements as in Table 1. From an economic perspective, the main ES are sustainable timber production and carbon storage.

Table 1: Annual Benefits of Sustainable Forest Management, AZ Costa Rica

Item	\$ ha^{-1} $year^{-1}$
Production Function	**\$75**
– Sustainable timber harvests	60
– Sustainable extraction of non-timber forest products	10
– Sustainable ecotourism	5
Regulatory Function	**\$105**
– Carbon uptake and storage*	105
Habitat Function	**\$20**
– Existence value*	10
– Biological prospecting (incl. Pharmaceutical value)*	10

Source: Bulte *et al*. (2002). * indicates an international spillover benefit.

The benefits of forest conservation that are external to the domestic country but spill over to other countries are considerable. Costa Rica is able to capture some of the rents associated with bioprospecting (through a deal between INBIO and the pharmaceutical company Merck), but that amounts to only a part of the estimate of the full rent of \$10 ha^{-1}. Table 1 suggests that more than 60% of the benefits of forest conservation do not materialize in Costa Rica itself – the country that has to bear the cost of forest conservation in terms of foregone agricultural production.

How much forest should be conserved?

Assume that Costa Rica ignores transboundary benefits. Then, the domestic benefits of sustainable forest management are between \$75-85 ha^{-1} yr^{-1}, depending on how much of the rents of bioprospecting Costa Rica is able to capture. By balancing the returns to agricultural expansion and forest conservation at the margin, the planner in Costa Rica should *deforest all land* for zero bioprospecting rents, but retain some 120,000 ha of forest for equal sharing with the pharmaceutical company. Two observations stand out. First, while returns to bioprospecting are modest, compared to some of the other items in Table 1, they make a large difference in terms of optimal land allocation. This is due to the fact that the function relating the shadow price of agricultural land to the area under cultivation is non-linear and virtually flat after 100,000 hectares of agricultural land (when additional deforestation is for extensive cattle ranching only). Second, the domestic optimum in the presence of bioprospecting rents is close to the current forest area in the region, which comprises of some 80,000–100,000 ha. In contrast, if Costa Rica is fully compensated for the external benefits of forest conservation, such that these benefits add up to \$200 ha^{-1} yr^{-1}, the economically optimal forest stock is 176,000 ha. From a global perspective, therefore, it is optimal to abandon some of the cattle ranches and let them convert back to forests.

4. Sources of market failure and related policies

Why do we generally not observe socially optimal levels of forest conservation in reality? Given our knowledge of the direct and underlying causes of deforestation, why do sovereign governments not implement policies and institutional structures to achieve such optimal levels? As we have indicated earlier, excessive deforestation is the consequence of various types of market failures. We now turn to a description of the main types of market failures and potential policy approaches to address them (section 4.1). Given the relative importance of international spillover effects highlighted earlier, our discussion about policies (section 4.2) will focus largely on this particular type of market failure. However, some policy issues related to other market failures are included in section 4.1.

4.1 Sources of market failure

Several sources of market failure in forest management can be distinguished. First, the lack of a complete set of forward markets, in particular, missing markets for forest services accruing beyond the local level may lead to regional and global externalities (i.e., spillover benefits such as carbon sequestration, watershed protection) not being considered in forest management decisions.

Other sources of market failure may induce inefficiencies in forest management even at the local level. A major one consists of improperly designed property right regimes and associated problems of tenure insecurity. Ideally, to provide incentives for efficient forest management, property rights should be universal (i.e. clearly defined for all scarce resources), exclusive (i.e. all benefits and costs from owning and using the resource should accrue to the owner), transferable from one owner to the other in a voluntary exchange, and enforceable (i.e., secure from involuntary seizure or encroachment by others) (Tietenberg, 1988). As will be discussed in more detail in section 5, property rights could in principle be allocated to private individuals, local communities and user groups, or the state. In all three cases, however, formal property regimes often degenerate into situations of *de facto* open access or at least a situation where enforcement is weak. This may be due to policy failure, but is also a consequence of the fact that forests exhibit inherent characteristics of common-pool resources (e.g., extraction of NTFPs is rival, but exclusion is often difficult and costly) and public goods (e.g., ecological services are often non-rival in consumption and non-excludable). It is well known that open access leads to an overexploitation of the forest resource as individual users do not take the effect of their use on the availability of the resource to other users into account, resulting in the so-called 'Tragedy of the Commons' described by Hardin (1968)[6]. Thus, open access may induce land races as farmers clear forest to prevent others from claiming the land (Angelsen 1995, 1999) or to benefit from land speculation (Clark et al., 1993). Tenure insecurity also contributes to favoring short-term benefits over long-run investments. In some cases, property rights are well-defined and enforced, but not transferable, reducing the incentives for forest owners to consider the impacts beyond their own lifespan.

A third source of market failure are market imperfections, including imperfect competition or imperfect capital and land markets. For example, lack of access to credit may prevent local forest users from using efficient technologies (e.g., improved cooking stoves) or sustainable practices (e.g., reforestation, plantations). This is aggravated where tenure insecurity prevents the use of forest land as collateral. As a consequence, local forest dwellers may end up in a vicious cycle of poverty and excessive resource extraction to satisfy subsistence needs (the so-called 'fuelwood trap') (Wunder, 2000). Similarly, land market imperfections may prevent farmers from fully internalizing long-run effects of their actions.

A fourth source of market failure related to the above is that private and social discount rates often differ (Tietenberg, 1988). For example, tenure insecurity, poverty, economic and political risks may cause individual farmers or local communities to discount the future more than a social planner. This is not to say that governments will necessarily discount optimally (and it should be mentioned that the 'socially optimal' discount rate is the object of much debate). For example, politicians may take their decisions on the basis of relatively short election cycles. Finally, market failure may also be caused by a lack of information or knowledge about local needs, ecological conditions, and environmentally friendly technologies. Frequently, availability of information is asymmetric, with local people likely to have better information about local conditions, and governments potentially having better access to scientific and technological knowledge (Platteau, 2003).

For some of the above sources of market failures, the definition of appropriate policies is in principle straightforward, but policy failure often prevents their adoption. Rather than correcting market imperfections, government intervention has often aggravated the existing incentives for excessive forest exploitation, e.g., through public road-building, settlement schemes, credit and fiscal incentives, and macroeconomic policies (Wunder, 2000).[7]

[6] The term 'Tragedy of the Commons' is really a misnomer as the situation described by Hardin is one of open access rather than common property.
[7] See also the chapter by Deacon and Mueller in this volume on the political economy of property rights formation.

As stated earlier, probably the most severe source of market failure is the fact that many forest services, particularly the ecological benefits accruing beyond the local level, are not marketed. For example, timber prices in most cases do not reflect the ecological costs of deforestation, leading to excessive logging. Similarly, land use decisions do not consider the ecological impacts of deforestation leading to excessive forest conversion into agriculture or other uses. Resolving this problem is particularly difficult for international spillover effects, as costs and benefits of alternative approaches are distributed very unevenly between individual countries, especially between developed and developing countries. We will turn to these issues in section 4.2.

4.2 International forest conservation policies

In the past, policies to address the problem of missing markets for spillover effects have mainly consisted of (i) command-and-control regulations dictating specific land uses, and (ii) remedial measures to reverse or reduce damages (Pagiola and Platais, 2002). Both approaches are problematic. Regulations face severe enforcement problems and impose high costs on poor land users relying on the forest for their livelihood (Pagiola and Platais, 2002; Sawhney and Engel, 2003). Moreover, in the context of international spillovers the scope for the command and control approach is limited in post-colonial times. Remedial measures are often imperfect and far more expensive than preventive measures (Pagiola and Platais, 2002). More recently, three main alternatives have emerged, and together they encompass the entire range from the carrot to the stick.

Turning to the stick first, within the International Tropical Timber Organization (ITTO), consumer and producer countries agreed to ban international trade in unsustainably harvested wood from tropical forests after the year 2000. Many tropical timber countries were opposed to this form of intervention, claiming that trade restrictions are unfair as temperate zone logging faces less stringent restrictions, and because many countries in the temperate zone have already deforested their lands in the past. Moreover, when sustainability requirements are too stringent, it was feared that sustainable forestry becomes less competitive as a land use option, and will be replaced by alternative types of land use (such as plantation agriculture). The claim that trade restrictions are detrimental for development is harder to evaluate. While banning trade might come at a considerable short-term cost for certain underdeveloped regions, it is evident that banning trade might raise welfare in exporting regions when harvesting takes place in a so-called second best setting (e.g. Brander and Taylor 1997). In light of abundant evidence of illegal logging (e.g. Kaimowitz 2003) and legal logging distorted through corruption and rent seeking (Ross 2001), trade bans on timber from unsustainable practices may be welfare improving for exporting countries – albeit obviously not in the interest of illegal loggers or corrupt politicians.

While some progress has been made to promote sustainable logging, it is clear that most of the forest areas in the world still do not meet the ITTO criteria. Worse yet, even if ITTO meets its stated objectives, it will not stop deforestation. Most tropical deforestation is not caused directly by logging. Logging might badly damage natural forests and compromise their ability to provide some of the functions discussed above, and it might "open up" closed forests and make them accessible for agriculturalists of various sorts. But Barbier *et al.* (1990) argue that only a small fraction of the deforested areas, mainly in Southeast Asia, can be directly linked to the international timber trade – for the political economy behind forest destruction by logging, refer to Ross (2001). Conversion for agriculture (be it by smallholders or cattle ranchers) is typically more important than timber harvesting (e.g., Myers 1994). To tackle these processes requires implementing policies that go beyond the forestry sector.[8] However, for 'Northern' countries, promoting trade measures that restrict the flow of Southern timber is relatively inexpensive. Moreover, it is clear that there are parties in the North that actually benefit from such measures.[9]

The second approach to forest conservation consists of *indirect* interventions.[10] These take the form of either re-directing labor and capital away from destructive activities (e.g., through agricultural intensification or the provision of alternative income sources), or of encouraging commercial activities that supply ecosystem services as joint products (e.g. ecotourism) (Ferraro and Simpson, 2002). At the international level, examples of indirect approaches include forest sector policies, which often occur through financing of projects in the South. During the 1990s this involved a substantial investment of some U.S.$ 2 billion. Where did the money go? The forestry assistance portfolio has shifted away from traditional forest sector development, forest plantations and agroforestry towards protected areas and conservation. A large share has been allocated to the protection of specific forests. In

[8] Indeed, when sustainable logging policies undermine the profitability of timber harvesting in the South, it could inadvertently promote deforestation by raising the relative profitability of other, more destructive land uses.
[9] Engel (2003) shows that a complete ban on timber from unsustainable logging, given costly monitoring, is optimal for the importing country only under the restrictive conditions of a corner solution. Where these conditions are not satisfied, bans may be motivated by protectionism.
[10] There are various reasons why forest conservation funds have been allocated to a sectoral forest approach, rather than to other uses where they might have been more effective. For example, there are vested sectoral and institutional interests. Technical forestry assistance provides jobs and funds for international experts, and investments in equipment may open up new markets for domestic industries. Political expediency might also have mattered – allocating funds to the forests sends an unambiguous warm-glow signal to the general public.

general these efforts have met with little success in reversing land use trends.[11] Much of the remaining funds were used to finance pilot projects that provide technical assistance to forest product harvesters and farmers, and localized short-term subsidies for sustainable forest management and sustainable agriculture. Another item that looms large on the budget are activities like preparation of plans and programs, conferences, seminars, public relations activities and information systems. While education and awareness building may be effective in inducing more positive attitudes towards conservation, it is not clear whether these will translate into changes in behaviour in the field (Sawhney and Engel, 2003).[12] Too often these activities are treated as substitutes rather than complements of economic and regulatory mechanisms that may influence actual behaviour (Kaimowitz 2000).

Most of the money has been allocated to either sectoral forestry/conservation approaches, or to rather symbolic activities. This allocation of funds has been largely unsuccessful in containing deforestation rates. Kaimowitz concludes *"it takes a hardy soul or a strong imagination to argue that forestry projects had much effect."* This assessment is consistent with evaluations of the international Tropical Forest Action Plan (TFAP, see Winterbottom 1990) and, more recently, World Bank lending in the 1990s (the World Bank 2000). This begs the question why past efforts have been unsuccessful.

The obvious response is that policies have not been used to directly address the root cause of deforestation: market failure. Very few of the underlying causes of deforestation summarized in section 2 (e.g., relative prices, infrastructure investments, credit policies) can effectively be tackled through sectoral forest policies – they require a multi-sector approach rather than a rigid focus on the forest. Moreover, as argued by Ferraro (2001), indirect interventions are often complex to implement and inflexible, and their effects are often hard to predict due to all sorts of context-dependent ambiguities – recall the complex effect of agricultural intensification on forest conversion. Indirect interventions also suffer from a mismatch between policies and temporal and spatial dimensions of ecosystem conservation.

The third policy approach that has been gaining attention recently consists of *direct* payments for environmental services (PES). This involves monetary transfers for transboundary ecological services. Ferraro and Simpson (2002) demonstrate that PES are likely to be far more cost-effective than indirect approaches. Moreover, given the ambiguous

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[11] Efforts to protect specific forests have typically been marginally successful. While there is evidence of somewhat less clearing in protected areas (e.g. Deininger and Minten 1996), most protected areas also suffer from serious encroachment. Those areas that do not are often just too remote to encroach upon.

[12] In some cases, indiscriminate increases in education and awareness building may even be counter-productive, as they reduce the potential of third-party actors (NGOs, government) to intervene in response to specific local deforestation threats (Engel and López, 2004).

evidence on the effects of agricultural intensification and alternative income provision outlined in section 2, the effectiveness of indirect approaches in combating deforestation remains unclear. In some cases, such approaches may even be counterproductive (López, 1998; Ferraro and Kiss, 2002; Ferraro, 2001; Kiss, 2003). We now review the PES approach in more detail.

4.3 Direct interventions: Payments for environmental services

PES are market-based mechanisms for forest conservation. Under this approach, NGOs, governments, or international donor agencies make periodic payments to individuals or groups that supply environmental services (e.g., local communities protecting forests), and these payments are conditional on the services supplied. In this way, externalities are at least partly internalized by local people in their resource management decisions. Moreover, compensating local resource users, which are characterized by a high incidence of poverty and strong dependence on forests for their livelihood, can help alleviate poverty and compensate users for the private benefits forgone from reducing extraction of forest products (Pagiola *et al.* 2002). PES include, for example, conservation concessions, payments for carbon sequestration under the Clean Development Mechanism, bio-prospecting, land leases and easements, or performance payments and tax relief. Such initiatives have been increasingly implemented in developed countries (e.g., United States, Australia, Germany) and developing countries (e.g., Costa Rica, Mexico, Chile, Guatemala, Madagascar, India). Box 2 describes a leading example of a PES system implemented in a developing country: the case of Costa Rica.

Recent evidence indicates that low-income farmers and communities are more likely to benefit from PES schemes when they have secure tenure rights, and when the schemes support not just pure conservation, but also other environmentally-friendly activities, such as sustainable forestry, agroforestry, or eco-tourism (Rosa et al., 2003). There are, however, important aspects of the implementation of direct conservation payments in developing countries, that still need to be better understood. First, property rights over forests in developing countries are often still weakly defined or poorly enforced. This raises the issue of how PES can assure that the forest will not be degraded by actors other than the contracting party under PES (e.g., industry, other communities). Second, the geographic and infrastructure characteristics of developing countries imply that monitoring of actual environmental services provided is very costly. This implies that there is a potential for moral hazard, which is reinforced by weak property rights and when dealing with groups of individuals such as local communities. Third, rent-seeking is an important problem, both by individuals within the community as well as by local governments. Thus, it is important to consider how an equitable distribution of PES can be assured. Providing environmental services in one area may lead to an increase in environmental degradation in other areas as activities are simply shifted geographically (the so-called *slippage*).[13]

Finally, an important issue concerns whether payments for ecological services should go to governments or NGOs and local communities. The impact of PES on the realization of conservation objectives depends on who controls the management and conversion of forests. When choices of local communities are an important determinant of the fate of forests, then PES must trickle down to the field level to affect behavior and be effective. Under such circumstances, implementing a PES system may align conservation and rural development objectives. In contrast, when national government policies are a key driver of deforestation, it may be more effective and efficient to transfer payments to the central level. This would provide an incentive to consider forests as a legitimate part of the national development portfolio, and gives the government maximum flexibility in choosing how to regulate this asset. Note that such transfers, based on the realized contribution to forest conservation, could result in a conflict between conservation and rural development objectives. This could happen when the government imposes additional restrictions on forest use by rural communities to protect the foreign transfer flow, but fails to plow these transfers back into rural development schemes.

[13] In addition, when demand for ecological services is downward sloping and if certain suppliers of these services have market power, there is an incentive to restrict supplies in order to raise prices or transfers (see Stahler 1996). PES could, in theory, be bad for conservation if this is the case.

Box 2: Paying for ecosystem services; the Costa Rica experience

Costa Rica has benefited from its forest resources as a means to attract foreign tourists. Tourism is now the second largest contributor to GDP. Because of this direct and important link between forest resources and income, Costa Rica has a strong incentive to protect its forest base. To this end, the country has introduced a 'market-based' approach, the so-called PSA system, by which thousands of landowners receive direct payments for the ES provided by the forests on their land. PSA is arguably the most elaborate PES system in the developing world, and it took years of policy debate and consensus building to prepare its implementation. Services included in PSA are not only domestic benefits (such as water protection and enhancing scenic beauty of the landscape) but, interestingly, also transboundary benefits (carbon sequestration, biodiversity conservation).

The programme aims to reach agreements with private landowners to cover an area of 100,000 hectares. Selected landowners may qualify for three different contracts: forest conservation contracts ($210/ha in 5 years); sustainable forest management contracts ($327/ha in 5 years); and reforestation contracts ($537/ha in 5 years). These numbers are somewhat lower than the estimates of the value of Costa Rican ES as summarized in Table 1, but nevertheless represent a significant first step towards closing the gap between the value of forest services and the income they generate.

The main source of funding for the system is a domestic fuel tax, which raises some $7-9 million annually (Rodriquez Zuniga 2003). Other important sources of funding for the programme are "Environmental Service Certificates" (issued for voluntary contributions by the private sector), agreements with hydro-electric companies, and support from the international community. The international community has placed a high degree of confidence in the Costa Rican system and institutional framework. For example, to support the programme and facilitate its implementation, the World Bank has provided a $32 million credit line and the Global Environmental Facility (GEF) has given an $8,3 million grant.

By bundling the various services and base payments on the aggregate value of four key services, the PSA provides "relatively high payments which are used to promote not only plantation forestry but also regeneration of secondary forests and other degraded landscapes" (Rodriquez Zuniga, 2003). There is evidence that the system is a success as forest cover in Costa Rica is now increasing.

5. Property right regimes and their relation to market imperfections and policy impacts

The degree to which market failures occur, and the difficulties associated with implementing different types of policy interventions, depend on the property right regime in place. Formally, four basic property right regimes can be distinguished: (i) open access, (ii) state property, (iii) common property, and (iv) private property. Mixed forms, such as co-management as a mixture of state and common property also exist.[14] White and Martin

[14] Another mixed form, frequently found in Africa, is 'communal ownership', where primary forests and uncultivated woodlands are owned by the local community and controlled by village authorities, while exclusive use rights over cultivated lands are assigned to individual community members (Otsuka and Place, 2001, ch.1).

(2002) estimate that 71 per cent of the forests in developing countries are still owned and administered by the government. Fourteen per cent are estimated to be owned by local communities and indigenous groups, and another eight per cent are publicly owned, but reserved for communities and indigenous groups. The remaining 7 per cent of forests are privately held (White and Martin, 2002). In a different study, Scherr et al. (2002) estimate that at least 25 per cent of developing country forests are owned or administered by low-income forest communities.

However, *de facto* property rights may differ from formal, *de jure* property rights regimes. Particularly, if property right enforcement is insufficient or ineffective, forests under state, common, or private property often degenerate into situations of *de facto* open access.

The consideration of property rights and their interlinkages with market failures and policy impacts is important because property right regimes have been subject to considerable change over the past decades. While forests have traditionally been considered state property in most countries, the last two decades have shown a strong tendency to either devolve rights and responsibilities over forest management to local communities (resulting in common property or co-management regimes) or private individuals (private property regime).

Property right regimes differ in their ability to reduce transaction costs (such as monitoring, enforcement, and information costs), to internalize externalities, and to deal with market imperfections. They also differ in their susceptibility to policy failure. Moreover, there may be important differences in distributive and social impacts between property right regimes. We now discuss the different regimes and the specific issues they involve.

5.1 Open Access

Clearly, open access (i.e., the complete absence of effective property rights) is the most problematic case to be considered. The inherent incentive for exploiting the resource before others do so is unlikely to be altered by any policy intervention aimed at reducing market failure (Otsuka and Place, 2001, ch. 1).

Moreover, open access (or generally situations of weak property rights) may induce various types of strategic behavior aggravating deforestation. First, if deforestation by one agent (e.g., the state or the local community) reduces the profitability of forest clearing to the other agent, each agent may aim to ‘squeeze the other’ by clearing more themselves, leading to higher overall deforestation (Angelsen, 2001). Second, if resource users have to invest in costly exclusion of potential encroachers (due to a lack of government enforcement or weak definition of property rights), they may decide to increase the intensity of resource exploitation in order to lower the returns from encroachment (Hotte, 2001, 2002).

In some cases property rights are made conditional on land clearing, adding a further incentive for excessive deforestation, both by those aiming to claim property rights (e.g., squatters) and those trying to prevent it (e.g., large landowners) (Alston et al., 1999).

Finally, the fact that *de facto* property rights are endogenous under open access regimes can lead to unexpected policy effects. For example, a rise in off-farm employment opportunities for local community members may increase the community's opportunity costs of fighting off commercial logging companies and thus increase deforestation (Engel and López, 2004). Similarly, a lower discount rate may in some cases increase deforestation by making it more costly for resource users to fight off potential encroachers (Hotte, 2001).[15]

5.2 State property

A large proportion of developing country forests are still under state property. An advantage of the state property regime is that the state is more likely to internalize local and regional externalities as compared to individual households or communities. Governments may also have better access to scientific and technological knowledge. Moreover, policies to internalize international externalities (such as PES) may be easier to handle with the state as a single partner in the transaction, rather than a large number of local communities or farmers.

In practice, however, state control has often failed to prevent the degradation of national forests because of the very high transaction costs and information problems associated with the design of effective usage rules, monitoring and enforcement at the local level (Arnold, 1998). Moreover, forest-dependent communities often have customary rights to the forest which cannot be ignored by the state. A lack of formalization of these rights under a state property regime induces a strong degree of tenure insecurity on part of local forest users.

As a consequence of these problems, state-owned forests frequently degenerate into open access resources, particularly in frontier areas that are located far from markets and government administrative centers (Hotte, 2001). There is increasing evidence showing that a large portion of forest conversion and degradation is associated with illegal activities, both by local communities or squatters and by commercial interests. For example, illegal logging was

[15] A decrease in the discount rate increases the value of a sustainable land use to the first settler as well as the potential contestant. If the latter effect is strong enough, the first settler may decide to mine the resource to avoid a costly conflict with the contestant (ibid).

estimated to account for a substantial portion of total log production in many countries, ranging from 90% in Cambodia to 34% in Ghana (Smith, 2002).[16]

Clarke et al. (1993) shows that governments may optimally tolerate some illegal logging in the face of monitoring and enforcement costs. The failure of governments to enforce property rights is, however, not only due to the costliness associated with monitoring and enforcement, but also with political-economy considerations and policy failure. This is highlighted by recent evidence that illegal activities have not decreased (and may even have increased) in the aftermath of decentralization reforms, although the transfer of forest management responsibilities to local governments should have reduced monitoring and enforcement costs. For example, decentralization in Indonesia has led to an increase in 'illegal logging', as district governments sanction timber extraction activities that would be considered illegal by the central government in order to generate local government taxes (Casson and Obidzinski, 2002).[17]

In general, reasons for policy failure to enforce property rights include the desire to relieve social pressure through settlement policies[18], strong lobbying potential of large logging companies, as well as rent-seeking and corruption on part of government officials. Indonesia, again, is a case in point. Illegal logging under president Suharto was associated with several conglomerates and individuals with close connections to central government elite and key military figures (Casson and Obidzinski, 2002). Moreover, the exclusion of local communities from logging benefits during the Suharto era led local people to assist companies in logging outside concession boundaries in return for salary or rents (ibid). The general move toward democracy and decentralization following the fall of Suharto has led local governments in the forest-rich provinces to tolerate illegal logging by local communities in times of economic crisis, and the withdrawal of army forces has led to an enforcement vacuum, often resulting in a situation of *de facto* open access. The informal timber sector, in some cases now legalized, is an important income source of district-level civilian and military bureaucracies (ibid).

Clearly, state incentives to monitor and to prevent illegal activities are crucial to the implementation of any policy proposal oriented to internalize international spillovers. As

[16] As the author notes these estimates should be taken as 'best guesses' only since derivation and year of estimation vary for the different country sources.
[17] Palmer (2001) estimates that the scale of illegal logging more than doubled during the period of the Asian economic crisis (1996-1998).
[18] Settlement schemes and colonization programmes in many countries have played the role of a social 'escape valve' to relieve discontent among the poor and avoid land reform policies in countries with high inequality in land distribution (Dorner and Thiesenhusen, as cited in Wunder, 2000, p. 44).

illegal logging is often a symptom of deeper structural problems in the forest sector, potential strategies for combating it include the strengthening of judicial systems and the rule of the law and restructuring forest industries such that processing capacity does not exceed supply (USAID, 2002). Some countries have recently implemented innovative approaches to combat illegal logging. For example, Ecuador has created a Green Surveillance team (*Vigilancia Verde*)—financed with the proceeds from auctioning off seized illegal timber—in which public and private sector institutions cooperate to monitor illegal operations (ibid). In Cambodia, several international organizations set up a Forest Crime Monitoring Unit aimed at increasing accountability, transparency, and enforcement through representatives in various government organizations and log tracking. The 1996 policy reforms in Bolivia empower private citizens to inspect forest operations and denounce illegal activities, and reduce rent-seeking opportunities by allocating concessions through public auctions and reducing forest officials' discretion in setting concession fees (ibid).

Whether the state will, however, adopt these or other measures to combat illegal logging and land conversion is essentially an issue of governance and corruption (see the chapter by Deacon and Mueller in this volume). The Indonesian case highlights the fact that the local government or agency in charge needs to see a long-run stake in the protection of forests as an asset. The case of decentralized protected area management in Bahia, Brazil, provides a positive example where decentralization and eco-tourism opportunities have led to both environmental and economic benefits from forest conservation (Oliveira, 2002).

5.3 Common property and Co-management

As a consequence of state failure in sustainable forest management in the past, shrinking national budgets, and a general trend towards decentralization and participatory approaches, many countries have recently started devolving—at least partially—the rights and responsibilities over forest management to local communities or user groups. As Edmonds (2002, p. 1) states: "…nearly every country in the world is experimenting with some form of 'Community Forestry'". Devolution may result in a common property regime or some form of co-management. A potential advantage is that the collective management rules, informal courts and sanctions established by user groups can provide a cost-effective alternative to government control (McKean, 1995). Moreover, local communities may have better information about local conditions. Compared to private property, communities may be better able to deal with local externalities, to exploit risk-sharing benefits from exploiting the forest resources jointly, and to provide a more equitable distribution of benefits. Ostrom

(1990) describes several empirical cases where communities have successfully managed common-pool resources. Edmonds (2002) provides econometric evidence that devolution of forests to local user groups in Nepal has reduced resource extraction by approximately 14 per cent, while citing evidence that deforestation and forest degradation had accelerated under previous state management.

Conceptual modeling and empirical evidence, however, clearly indicate that community-based forest management is not without problems and risks. First, communities, just as individual households, have no incentive *per se* to internalize regional or international externalities. Implementing policies to correct this market failure, e.g. through PES, are however further complicated when dealing with local communities rather than the state or individual farmers, including potential problems of free-riding and moral hazard.[19]

This is related to the second potential problem with community-based management: It is naïve to think of local communities as homogenous groups, automatically acting in the interest of the whole. For example, evidence from Indonesia shows that, given rights to negotiate directly with companies over logging agreements, local communities often sell off the forest for short-term financial benefits at high environmental and social costs (Barr et al., 2001; Casson and Obidzinski, 2002; Engel et al., 2003). This is fostered by the fact that devolution is often incomplete in the sense that the state retains some rights over the forest or share of the benefits, leaving communities with reduced or uncertain incentives to consider the long-run effects of their actions. Moreover, achieving even a locally efficient level of resource management requires collective action on part of individual community members, i.e. their ability to agree on and enforce a cooperative and efficient set of access and use rules (Ostrom, 1990). Otherwise, common property, similar to open access, would result in overexploitation as each individual does not consider the impact of his action on resource availability to other users. Much literature has focused on the factors favoring the success of common property regimes. Small group size, social and cultural homogeneity, problem severity, high existing social capital, consistent impacts[20], low discount rates, and low transaction costs are widely seen as conducive for collective action, while the effect of economic heterogeneity (e.g., wealth and asset distribution) remains much debated.[21]

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[19] PES to local communities have been implemented, for example, in Costa Rica, Mexico, Guatemala, and Chile. They are also under development in Indonesia.

[20] 'Consistent impacts' refers to a situation where most individuals will be affected in similar ways by the proposed management changes.

[21] Baland and Platteau (1996) and Agrawal (2001) provide detailed reviews of this literature. Classic references include Runge (1986), Wade (1988), Ostrom (1990), Seabright (1993), Bardhan (1993a/b), Baland and Platteau (1997a/b, 1998). See also Mansuri and Rao (2003) for a detailed review of the evidence on community-based and community-driven development in a wider context.

Third, it has been shown that devolution may lead to rent-seeking activities by community elites and prevent the state from exercising an important role in assuring the inclusion of marginalized groups (Abraham and Platteau, 2002; Agrawal and Ostrom, 2002; Platteau and Gaspart, 2003; Platteau, 2003). There is empirical evidence that more powerful actors in the communities manipulate devolution outcomes in their own interest (Shackleton et al., 2002), at least where the poor are not empowered enough to oppose pressures from the local elite (Platteau, 2003). This is highlighted by empirical evidence from India and Nepal, both countries with major nationwide devolution initiatives. Agarwal (2001) discusses how seemingly participatory institutions often exclude significant sections of the community, such as women. Kumar (2002) shows for the state of Jharkhand in India that wealthier sections of the communities have benefited from the Joint Forest Management (JFM) program at the expense of the poor. Given a share in total benefits from state timber extraction, community rules now favor long-run timber benefits through forest closure and plantations of high-value species. As a consequence, poor forest-dependant households are marginalized as they suffer from the reduced availability of and access to NTFPs (ibid). One should be careful, however, to generalize these results as they are likely to depend on initial forest conditions and the type of forest. For Nepal, Karmacharya et al. (2003) found that some communities under community forestry programs have been successful in creating specific pro-poor rules and incentives. By contrast, the parallel program of leasehold forestry which explicitly assigns rights over degraded forest only to groups of poor households has led to serious enforcement problems due to the lack of recognition of these rights by other community members (ibid). The authors thus conclude that the government should share information about existing pro-poor provisions and encourage user groups to adopt similar rules.

The more favourable equity outcomes in the Nepalese case as compared to India, may be due to the fact that timber production for revenue plays less of a role in Nepal. Where community-based forest management involves financial transfers from donor agencies, NGOs, or government, sequential and conditional release of funds may be a useful approach to discipline local leaders (Platteau, 2003). However, for this approach to be effective, funding or implementing agencies need to cooperate to avoid competition among themselves to the benefit of local leaders (ibid). Among other things, this would require the systematic reporting of cases of failure, which stands in contrast to current practice by funding agencies (ibid).

The case of India also highlights the mixed evidence on the success of community-based forest management. Some studies have shown improvements in outcomes such as increased yields of timber, NTFP, fuelwood, and fodder (Joshi, 1999; Khare et al., 2000;

Ballabh et al., 2002). Others indicate a lack of control and management of forests by the communities, despite the fact that communities have *de jure* rights over the forest. For instance, Jodha's (1986) study of 82 villages in the dry region of the country revealed that not a single village was using control measures such as grazing taxes or penalties for violations of forest use rules. The variety in outcomes of JFM may partly be due to the fact that the degree to which specific rights and benefits were actually devolved from the forest department to local communities differs significantly across states (Damodaran and Engel, 2003). Cynics see JFM as just another way to extend forest department control to new areas as forest management committees under JFM are mostly still controlled by government staff (Sundar, 2001).

Co-management is often seen as a desirable combination of the advantages of state and common property. The above discussion indicates that the degree to which communities should be given rights and responsibilities over forest management depends on local conditions. Community involvement is crucial where there is a prevalence of indigenous and forest-dependent population, where natural resources are located in remote areas and are locally specific and diverse, and where there is a lack of state funds and monitoring and enforcement capacity. State involvement is crucial where externalities are important, marginalization of groups within communities is an issue, and where resource and user characteristics are such that the expected ability for community collective action is low. To avoid disincentive effects that are common in present systems, co-management would probably work best if the state took on the role of overall planning and assistance, and potentially put some restrictions to community activities, while the communities are allowed to decide freely and obtain full benefits within the areas and limits set by the state (Engel, 2004). At least three necessary conditions for effective co-management emerge from a review of the literature (Ribot, 2002; Larson and Ribot, 2004): (i) secure and well-defined property rights; (ii) transfer of appropriate and sufficient powers to communities, and (iii) downwardly accountable and representative local institutions. Table 2 also summarizes important roles stressed in the literature for the state as well as the international community (donor agencies, NGOs, etc.) in a co-management system.

Table 2: Important roles for various actors in common-pool resource management

THE STATE
o Setting national environmental priorities and standards (boundaries to community management, zoning)
o Deal with national and regional externalities
o Provide a legal framework which enables communities 'to obtain legally enforceable recognition of their rights and to call upon the state as an enforcer of last resort'
o Legal backing for community-established use and access rules
o Provide conflict resolution mechanisms
o Provide technical assistance, monitoring equipment, scientific information, awareness building
o Provide economic incentives for conservation (e.g., poverty alleviation, payments for environmental services)
o Provide information on best practices, experiences of other communities
o Assure inclusion of marginalized groups, promote pro-poor rules
o Promote grassroots participation and downward accountability (e.g. through popular elections, forums for discussions and negotiations, mandated financial reports, social audits or vigilance committees)
INTERNATIONAL COMMUNITY/DONOR AGENCIES/ NGOS
o Implement mechanisms to translate international externalities into economic incentives for sustainable resource management and conservation in developing countries
o Support state in its functions
o Support horizontal coordination between community organizations (e.g. regional forest user associations) to address broader-scale problems and increase bargaining power vs. the state.
o Support good governance (rule of law, secure land tenure, democratic elections, government accountability)
o Help overcome 'culture of distrust' between communities and the state, build community confidence in its collective action ability, build up or adapt local institutional arrangements for resource management
o Capacity building at local level (financial and administrative management, technical skills, problem-solving)
o Awareness building at local level (providing information on rights, success stories)
o Promote the participation of marginalized groups (e.g. by helping them to raise their voice in defense of their interests and demand transparency and accountability)

Source: Baland and Platteau (1996), Ribot (2002), Larson and Ribot (2004)

5.4 Private property

Some countries, e.g., China and Vietnam, have recently started privatization programs for formerly state-owned forests. Private ownership as an alternative to state or common property regimes has long been promoted by economists who argue that privatization, by assigning all benefits and costs for a particular forest area to one individual, yields the optimal incentives for an efficient resource management.[22] The argument that private property rights satisfying the criteria of universality, exclusivity, and transferability will yield efficient market equilibria, however, relies crucially on four important assumptions: a) enforcement costs are nil, b) property rights are well defined, c) markets are perfect, and d)

[22] See, for example, Demsetz (1967), Hardin (1968), Posner (1977), Anderson and Hill (1977).

markets are competitive (Baland and Platteau, 1996, p. 37). In the context of developing country forests these conditions are usually not satisfied.

First, the prime criticism of the private property rights school is that it underestimates the transaction costs involved in monitoring and enforcing private property rights (Baland and Platteau, 1996). the very characteristics of common-pool resources that make exclusion of potential users difficult may make it even more difficult to achieve a degree of separation, exclusion and protection necessary to privatize it (Arnold, 1998). Unsecured *de jure* rights, which are not enforced or prohibitively costly to enforce, may not improve tenure security. In developing countries, insecurity, high transaction costs, poor, partial and arbitrary enforcement of rights due to weak judiciary and constitutional laws and lack of infrastructure can thus seriously constrain the efficiency of individual property rights, especially if those rights do not enjoy the support of custom (Feder and Feeny, 1991).

Second, private property rights do not account for externality effects, neither at the global and regional level, nor at the local level (contrary to common property rights which may at least internalize local externalities such as soil erosion). Third, forests often need to be managed in their entirety in order to maintain their ecological functioning. Fourth, common use of a forest can reduce the individual users' risks in areas where the location of the most productive zones can vary from year to year.[23] Finally, privatization usually implies that some former users are excluded, which may have undesirable equity and poverty implications. Assuming that privatization actually does lead to efficiency gains, Weitzman (1974) shows that if privatization leads to resource ownership by an outsider or if the state sells off competitively the rights to the resource, former resource users (e.g., traditional communities) always lose despite the fact that they may become wage earners and wages increase as a consequence of the efficiency gains. If, on the other hand, former users get their rights recognized and are compensated for losses incurred from privatization, everybody can gain from privatization; but empirically this is often not the case due to political economy considerations and information problems (Baland and Platteau, 1996). Otsuka and Place (2001) present evidence from several African and Asian countries indicating that land distribution under common and communal property regimes tended to be more equitable than in private property regimes.

The optimal property regime may also depend on the type of forest. For example, Otsuka and Place (2001) conclude that common property regimes are effective when NTFPs are the predominant forest products. They argue that management of timber forests (where

[23] This is the case, for example, of woodland in arid areas.

protection is less costly, but management intensity is high) is more efficient under private land tenure than under common property regimes. However, where protection is costly (e.g., due to a threat from grazing), a combination of private and common property system may be optimal, where management is carried out individually or by a centralized management committee, and protection is carried out communally (ibid, p. 48).

6. Conclusion and Discussion

In this chapter we have highlighted the key role of market failure in tropical forest conservation. The mismatch between costs and benefits of forest conservation, both in space and in time, implies that too much forest area will be converted to other uses. Policy failure is a compounding factor, making matters even worse.

We have explored the various policy options that exist for the rest of the world to address market failure. There appears to be a slow trend towards payments for ecological services, a development which we believe is encouraging as it tackles the market failure problem directly, which is likely to be both efficient and cost-effective. North-South transfers also reduce distributional inequality and may help promoting rural development. A recent review found almost 300 examples of marked-based approaches to forest conservation, mainly in Latin America and the Caribbean (Landell-Mills and Porras 2002). This list is growing, which offers hope that cost-effective and efficient conservation of valuable natural capital may be within reach. This tendency will offer increasing opportunities for the international community to 'put their money where their mouth is', and reach fair agreements with selected and accountable governments in developing countries. The great challenge for forest conservation in the future may be to move beyond local cases and implement a consistent plan of action that encompasses many countries.

However, while forest conservation expenditures have increased considerably since the 1970s (the total amount involved in the 1990s amounted to some $2 billion – or an average of $200 million per year), it appears as if actual North-South monetary transfers still lag behind the value of the South-North spillover benefit by a few orders of magnitude. It appears that parties in the North simply prefer to free ride on conservation in the South – an attitude that would not only be unfair, but also inefficient and unsustainable in the long run. The suggestion of "willingness to free ride" is somewhat reinforced when we take the evidence into account that consumers in the North are unwilling to pay a premium for sustainably produced tropical timber, or for biodiversity-friendly shade-grown coffee from forest regions. None of this is surprising given the public good nature of most forest services.

Not only may the North free ride on the South, but also each individual country in the North has an incentive to free ride on the others' actions. Given the public good nature, it is unlikely that a true market for international spillover effects will evolve in the absence of international agreements assigning clear property rights and implementing supra-national enforcement mechanisms. The discussion on carbon trade is encouraging in this regard (but difficulties in including all relevant parties to sign the Kyoto protocol is illustrative). As long as such markets are not in place, however, PES are likely to take the form of bilateral agreements. The theory of public goods clearly predicts that this will lead to an underprovision of forest services. PES are essentially forms of bargaining. The Coase theorem, however, states that such bargaining emerges where property rights are well-defined and transaction costs are small. Both conditions are not satisfied in an international context. This may also explain why large PES schemes such as the Costa Rican scheme has emerged from a national rather than international initiative. Again, resolving these issues on an international level will require cooperation and coordination between countries, particularly within the North.

There exists another, more hopeful perspective. The majority of the current efforts to provide direct compensation for forest services deal with local or regional issues – mainly watershed management. While one can be sceptical about the prospects of fully incorporating transboundary services in the future, it could also be argued that the current phase of incomplete compensation for such services is necessary to build the appropriate institutions. Not all countries are like Costa Rica, where payments immediately trickle down to landowners and affect their decision-making. For many countries, this requires considerable challenges in terms of public governance, financial administration and political will.[24] Landell-Mills and Porras (2002) emphasize the importance of secure land tenure, good governance and a strong legal framework to provide an 'enabling environment' in which market-based incentives can prosper.

We have also discussed how market failures and policy impacts depend on the property rights regime in place, an aspect that has been subject to considerable change over the past decades. It appears clear that secure and well-defined property rights are a crucial precondition for any policy to be effective in reducing deforestation. Which property rights regime is appropriate, however, depends on ecological conditions, user characteristics, dominant extractive activities (e.g., timber vs. NTFPs), and political economy considerations.

[24] The Costa Rica example is of course not perfect. Among other things, it has been criticized because of its emphasis on forest conservation (as opposed to providing development opportunities for the rural poor).

It is now generally agreed that pure state property regimes have performed very poorly in the past, and that it is impossible in a developing country context to simply protect the forest from the local people depending on them for their livelihood. The move towards devolution and the participation of local people in forest management is certainly a move in the right direction. However, common or private property regimes are not without serious problems and risks. If properly designed, co-management regimes could benefit from the comparative advantages of both governments and local communities in managing local forests (Knox and Meinzen-Dick, 2001). However, in reality, it seems that devolution of rights has often not kept pace with devolution of responsibilities, and that government agencies—while increasingly adopting the rhetoric of devolution and participatory approaches—are often reluctant to give up substantial powers, resulting in half-hearted policy change with potentially counter-productive effects.

To improve current systems will also require a better understanding of past experiences. Particularly, the literature on government-initiated community institutions and the outcomes of privatization is limited. As Agrawal (2001) states, we need to go beyond the examination of individual case studies towards more systematic, large-sample studies. Two issues that need particular attention appear to be the importance of fund-transfers for the sustainability of effective community-based forest management institutions and the related incidence of rent-seeking at the national vs. local level (both for local governments and local community elites). Also, current analyses of 'community-driven development', 'community-based management', 'devolution', and 'participation', include many very different types of institutional settings. Therefore, not only do we need a better understanding of determinants of success within a given setting, but furthermore a comparison across projects and countries to improve our understanding of the impact of the overall institutional framework conditions (e.g., the degree to which rights and responsibilities are shared between state and communities under co-management regimes) is required.

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