Title: *Constructing Markets for Ecosystem Services: limitations of development interventions and a role for conservation performance payments*

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Abstract:

Conservation biologists, policy makers, and citizens have identified the protection of native ecosystems in low-income nations as a global social objective. Among the more popular initiatives to achieve this objective is the use of development interventions in the peripheral areas of endangered ecosystems. Such interventions *indirectly* provide desirable ecosystem services through two mechanisms: (1) by re-directing labor and capital away from activities that degrade ecosystems (e.g., agricultural intensification); and (2) by encouraging commercial activities that produce ecosystem services as joint products (e.g., ecotourism).

By examining the economics of such interventions and the available empirical evidence, I argue that development interventions are hindered by: (1) the indirect and ambiguous conservation incentives that they generate; (2) the complexity of their implementation; and (3) their lack of conformity with the temporal and spatial dimensions of ecosystem conservation objectives.

In contrast, paying individuals or communities *directly* for conservation performance may be a simpler and more effective approach. In recent years, there has been widespread experimentation with *contracting* approaches to ecosystem conservation. Contracting approaches create markets through which individuals who provide ecosystem services can benefit from their efforts.

An examination of current contracting initiatives indicates that they have many advantages over more popular indirect development interventions. Conservation contracting can: (1) reduce the set of critical parameters that practitioners must affect in order to achieve conservation goals; (2) permit more precise targeting and more rapid adaptation over time; and (3) strengthen the links between individual well-being, individual actions, and habitat conservation, thus creating a local stake in ecosystem protection. In situations where performance payments are unlikely to work, indirect development interventions are also unlikely to work. Thus despite the potential barriers to developing a system of conservation contracts in low-income nations, the analysis suggests that performance payments have the potential to vastly improve the way in which ecosystems are conserved in low-income nations.

## Introduction[1]

Imagine that you live in a house that needs no air conditioning because the trees on your neighbor'sproperty provide shade to cool your home. Recently, however, a new person moved into your neighbor's house. He wants to cut down the trees because he has installed solar panels to reduce his electricity bills. Cutting down the trees will increase the efficiency of the panels, but will require you to install air conditioning and pay much higher electricity bills. Your visiting uncle, who consults for integrated conservation and development projects, suggests a plan for ensuring that your neighbor's trees remain standing. He suggests that you create alternative employment and investment opportunities in which your neighbor's returns to labor and capital are so high that he will not want to invest time or money in cutting down his trees.

You may, however, decide that it is easier, and probably cheaper, to simply offer your neighbor an annual payment to leave the trees standing. The payment would have to be large enough to compensate your neighbor for the foregone reductions in his electrical bills, but it would probably be far less than the cost of occupying your neighbor's resources in other activities. Moreover, the probability that the trees will remain standing is higher.

Paying your neighbor to leave his trees standing because they provide a valuable service would strike few people as misguided. In low-income nations, however, citizens and governments interested in habitat and biodiversity conservation have adopted a less direct and more complex approach. Rather than make explicit payments, they use field-based project and policy interventions to transform local and regional economies in ways that encourage individuals to invest in activities that do not lead to habitat or biodiversity loss. They propose, in effect, to guide the economic development process toward paths that are compatible with ecosystem protection.

The premise underlying these interventions is sound: if residents near a threatened ecosystem are the principal agents of change, their behavior must change to protect the ecosystem. Even if residents are not the principal agents of change, they are often in the best position to protect the ecosystem and so influencing their behavior is important. Problems arise, however, when one examines the links between ecosystem conservation and the myriad interventions proposed by conservation practitioners (e.g., agroforestry, ecotourism). The next section explores the logical problems associated with using development initiatives to

address the loss of *intact ecosystems (habitat)* and the concomitant loss of *biodiversity*. The final two sections introduce a more direct approach that depends on explicit payments tied to conservation results.

The purpose of this paper is to draw attention to and generate discussion about a system of direct payments for achieving ecosystem conservation objectives in low-income nations. Emphasis is placed on the positive aspects of direct payments. Problems associated with systems based on direct payments are not ignored, however, and I hope that this paper can serve as a foundation for examining direct payment systems more thoroughly.

**Development Interventions as Means to Achieve Ecosystem Conservation**

In this section, I focus on *field-based* interventions, such as technology transfers, that target *individuals* living near an ecosystem. Broader policy interventions, however, are clearly important in low-income countries. Ecosystem degradation is often stimulated by road building in remote areas, by direct and indirect subsidies for production activities, and by policies that encourage farmers to clear land in order to avoid taxes or gain property rights. Changes in these policies are thus a necessary condition for ecosystem conservation, but they are unlikely to be sufficient. In the best cases, broad policy changes will reduce pressures on ecosystems by slowing conversion, but they are unlikely to remove all of the individual-level incentives for converting habitat to other uses. Habitat conservation will typically require more precise, field-level interventions.

Below, I identify three principal problems associated with using development interventions to protect ecosystems. First, given the complexity of development interventions and the temporal and spatial scales at which conservation objectives must be achieved, field practitioners are forced to spread their resources over myriad tasks that often have no effect on conservation-related household behavior. Second, when practitioners do manage to have an effect, it is often an undesirable effect from a conservation perspective. Third, even if practitioners generate a desirable effect, they often have difficulty sustaining it because the effect depends on market conditions that change frequently.

*Barriers to Change*

Experience with development interventions over the last four decades indicates that simply raising standards of living and encouraging economic growth is a major undertaking in many countries (World Bank 1988; Porter et al. 1991). Advocates of development-based conservation interventions propose a much more difficult task. They propose, in effect, to guide or control the development process so that specific behavioral changes will occur and precise conservation objectives will be achieved. They are attempting not simply to effect change, but to control the precise evolution of the change.

A quick review of probability theory illustrates a key problem associated with development interventions aimed at promoting habitat conservation. The joint probability of two independent events, each of which has a 0.75 probability of occurring, is $(0.75)^2$ or .56. Consider a "project" that requires the successful completion of twenty independent steps or tasks. Suppose each of these steps is likely to be completed; i.e., three out of four times. By the same logic as above, the probability that the overall project will succeed is 0.003; i.e., essentially there is no chance of success. While this example does not match perfectly the implementation of development interventions, the lesson derived is still applicable: if one wants a successful intervention, one must keep the effort simple and keep it focused on likely events.

Development-based conservation interventions, however, often exhibit the exact opposite characteristics. For example, the annual work plans for five integrated conservation and development projects in Madagascar listed on average forty key activities, not including administrative tasks (CI 1995; PPNR 1995; VITA et al. 1995; WWF 1995; CI et al. 1996). Moreover, many of the tasks listed (e.g., intensify agriculture, develop community institutions) were likely to involve many sub-tasks in order to be successful.

Given the immediacy of conservation objectives, practitioners must also achieve results quickly. Development interventions, however, rarely produce significant transformations of economies and individual behavior in the short run. New technologies, markets, and attitudes take many years to develop and slowly work their way through societies.[2]

In order to reconcile the short-term immediacy of habitat conservation objectives with the slow pace of social change, conservation practitioners resort to two approaches, often in

combination. The first approach is to regulate by force in the short run, while practitioners wait for changes in resource use incentives to materialize. Successful development interventions, however, depend on trust and cooperation between residents and outside technicians. Trust is difficult to engender in the presence of repressive force. The second approach involves spending large amounts of money and resources to introduce new technologies, infrastructure, and attitudes quickly. The history of development interventions in low-income nations, however, speaks for itself: attempts to introduce multiple technological, institutional, and attitudinal changes simultaneously typically fail (World Bank 1988; Porter et al. 1991).

The fundamental tension between the time horizons of conservation and development objectives is further exacerbated by differences between the appropriate spatial scales at which conservation objectives and development interventions are realized. Ecosystems are often very large and encompass many biological and cultural zones. Thus the effort to conserve them must be accomplished at a landscape scale. Development initiatives, however, are context specific and often best begun on a small-scale (Bunch 1982) or with a narrowly defined focus (World Bank 1988). When practitioners quickly introduce new technologies, markets, and attitudes at large scales, they spread their resources thinly over a large territory, thereby diluting their effect.

*Barriers to Desirable Change*

Even if conservation practitioners are able to effect changes in a local economy, their interventions may not alter the incentives that prompt rural residents to degrade habitat. Rather than serving as *substitutes* for ecosystem-degrading activities, the new technologies and employment opportunities introduced by conservation practitioners are often *complements* (Ferraro et al. 1997). In other words, residents adopt the new technologies or employment opportunities (e.g., animal husbandry) *and* continue to engage in activities that threaten ecosystems (e.g., hunting).

Moreover, the new technologies or employment opportunities can exacerbate habitat loss (see *Appendix A* for an analytical illustration). For example, contrary to the dominant hypothesis of many conservation and development projects, agricultural intensification will *not* necessarily take pressure off of native ecosystems. In fact, some studies suggest the

opposite: decreases in input prices (Ozorio de Almeida & Campari 1995; Lewandrowski 1997) and increases in productivity (Wiersum 1986; El Nagheeb 1990; Barraclough & Ghimire 1995; Kaimowitz & Angelsen 1998:51; Foster et al. 1999; Angelsen 1999) in low-income nations are associated with increases in the area of land under agriculture. A recent review of deforestation analyses found that the relationship between intensification and deforestation was indeterminate (Kaimowitz & Angelsen 1998).

Successful agricultural development interventions raise household incomes. More income permits farmers to purchase more labor and capital with which to further expand their activities. The needs of most people are not finite, particularly those of poor farmers. If farmers can be better off by expanding new or old technologies into intact ecosystems, they will do so. An increase in the returns to agriculture can therefore be equivalent to an increase in the opportunity costs of conservation. In such cases, conflicts between local residents and conservation practitioners will grow with increases in agricultural productivity.

Successful agricultural development interventions in rural areas typically also require improvements in transportation and market infrastructure. A review of deforestation analyses (Kaimowitz & Angelsen 1998), however, found that many studies link deforestation to proximity and quality of transport routes and markets. Better infrastructure can make pro-conservation activities more profitable, but it can also make other activities more profitable as well.

The introduction of better infrastructure and new livelihood opportunities also tends to encourage immigration into a region. Thus, even if one could implement a labor-absorbing development strategy to promote habitat conservation, the pool of labor may simply expand and render the strategy ineffective. Unless current residents have a direct incentive and the ability to protect the ecosystem from conversion, the entire "agricultural sector" near the ecosystem will expand. This phenomenon has been noted in case studies and general equilibrium analyses (e.g., Jones 1989; Elahl & Khushalani 1990; Coxhead & Jayasuriya 1994; Ferraro et al. 1997; Oates 1999).

A paradox seems to exist. Stagnation in the agricultural sector can put pressure on forests as farmers extensify their production and the landless migrate to the forest margins. On the other hand, agricultural profitability threatens forests by increasing incentives to clear land for cultivation. The paradox is resolved by recognizing that, at the most fundamental

level, the profitability of agriculture, no matter how marginal, drives habitat conversion. Therefore, only the profitability of conservation can arrest it.

To increase the profitability of protecting ecosystems, practitioners have turned to another popular approach: market-based initiatives, such as selective timber logging or nontimber forest product extraction, that raise the local value of intact ecosystems. Experience to date with such initiatives, however, indicates that success is likely only under limited conditions (Campbell et al. 1999; Salafsky et al. 1999). The task of turning remote rural residents into eco-entrepreneurs is complex, and most projects yield too few benefits, for too few people, to compete with activities that lead to habitat conversion (Browder 1992; Richards 1993; Lawrence et al. 1995; Smith 1996). Attempts to increase the benefits from ecosystem use often lead to the degradation or simplification of the ecosystem (Freese 1997). Even low intensity, subsistence (Redford 1992) or commercial (Howard et al. 1996; Peres 1999) activities can lead to the same outcome. Moreover, the scientific data required to determine appropriate extraction levels may be expensive to gather (Freese 1997). Other authors have also noted problems related to the sustainability of extractive initiatives (Tewari & Campbell 1996; Barrett & Arcese 1998) and the inefficiencies of subsidies that are often required to make eco-activities profitable (Simpson & Sedjo 1996).

Despite the theoretical appeal of interventions oriented toward increasing the local value of intact ecosystems, the practical implementations to date have many shortcomings: they often fail to match the benefits generated by ecosystem conversion, they can lead to undesirable ecosystem simplification, and they require significant resources to implement, monitor, and sustain.

*Barriers to Sustaining Desirable Change*

Even if practitioners overcome the problems outlined above, an important obstacle remains: how to *maintain* the created system of incentives for habitat protection. Development-based conservation approaches appear to assume implicitly that one can intervene in an area, transform the local or regional economy, exit, and then watch as the transformed system rolls along in perpetuity. Societies, their economies, and their environments, however, are never static. Prices change, roads degrade, new pests develop, and new information arrives. Development-based initiatives will therefore inevitably require

repeated interventions over time. In the long run, such approaches are likely to be extremely expensive, even if they are successful in the short run.

**A Role for Conservation Performance Payments**

Development interventions will not make ecosystem protection optimal for rural residents in many areas of the world. As argued above, the links that development interventions create between individual well-being and habitat conservation are often vague and indirect, or simply nonexistent. Thus conservation practitioners find that creating an appropriate set of incentives and maintaining these incentives over time is difficult.

Despite the difficulties in using development interventions to promote habitat protection, conservation practitioners should not abandon attempts to change field-level incentives. As Laarman (1995:53) argues, the challenge is to test and ultimately implement interventions superior to our current efforts, not to discard the principles of intervention. Given the discussion above, ideal interventions should have the following characteristics:

1) they are relatively simple, in the sense that they allow practitioners to focus their energy on a few activities with high probabilities of success;
2) they achieve conservation objectives in the short-term and the long-term;
3) they achieve conservation objectives at the scale of ecosystems;
4) they provide clear, direct incentives for residents to actively protect habitat;
5) they deter entry/immigration; and
6) they reduce the social and political conflicts over resource allocation that often endanger ecosystem survival.

To design an intervention possessing these six characteristics, practitioners may consider an *International Habitat Reserve Program (IHRP).* An IHRP is a system of institutional arrangements that facilitates *conservation contracting* between international or national actors and individuals or groups that supply ecosystem services. The contracts specify that the outside agents will make periodic *performance payments* to local actors if a targeted ecosystem remains intact or if target levels of wildlife are found in the ecosystem.

The notion of compensating people for their role in maintaining resources that have a global value is not new (Barbier & Rauscher 1995; Swanson 1995; Simpson & Sedjo 1996). Direct compensation schemes for individuals living near protected ecosystems are rare, however, because there are serious obstacles to designing an effective scheme (Simpson & Sedjo 1996; Ferraro & Kramer 1997). Practitioners must deal with strategic behavior by recipients, the complexity of institutional design, conflicts over property rights, and potentially high costs of implementation. Practitioners can, however, avoid these problems if they learn from the many successful examples of initiatives that pay individuals or groups for conservation performance.

The best known conservation payment initiatives are the agricultural land diversion programs of high-income nations. In Europe, fourteen nations spent an estimated $11 billion (1993-97) to divert well over 20 million hectares into long-term set-aside and forestry contracts (OECD 1997). In the United States, the Conservation Reserve Program (CRP) spends about $1.5 billion annually to contract for 12 - 15 million hectares.[3]

These conservation contracting programs account for only a few percent of agricultural support budgets, but they are among the fastest growing payments to farmers in high-income nations (OECD:14). Their dramatic growth is partly due to their popularity among various stakeholders (OECD:20) and the opportunities they afford for flexible targeting and adjustment to local conditions (OECD:48). Moreover, payments for enhancing the supply of environmental goods and services are likely to be one of the few government transfers to rural farmers that global trade organizations, like the WTO, will countenance (Potter & Ervin 1999).

Nongovernmental organizations (NGOs) have also developed innovative direct payment approaches. The Delta Waterfowl Foundation, for example, has an “adopt-a-pothole” program that pays prairie farmers who protect nesting areas for ducks (Delta Waterfowl Foundation 2000). The Defenders of Wildlife have a program that rewards landowners for occupied wolf dens on their property (Defenders of Wildlife 2000).

Although rare outside of high-income countries, direct payment systems can also be found in the tropics. In the last four years, Costa Ricans have created institutional mechanisms through which local, national, and international beneficiaries of ecosystem services compensate those who protect ecosystems (Castro et al. 1998; Calvo & Navarrete

1999). Costa Rica's 1996 Forestry Law (no. 7575) explicitly recognizes four ecosystem services: carbon fixation and sequestration, hydrological services, biodiversity protection, and scenic beauty. The law gives landowners the opportunity to be compensated for the provision of these services.

Costa Rican practitioners have identified sources of financing and have developed rules for allocating available funds. Funds are currently allocated through the National Forestry Financial Fund (FONAFIFO), which works directly with landowners and indirectly through third-party intermediaries (e.g., NGOs). FONAFIFO raises money from international donors and national sources, such as a fuel tax and payments made by hydroelectric plants. FONAFIFO then distributes the money through contractual arrangements with private individuals and groups.

FONAFIFO establishes contracts for three land use categories: reforestation, sustainable forest management, and forest preservation. The forest preservation category is the most common contract. Each category is associated with a fixed annual payment per hectare. Regional conservation agents and third-party NGOs identify potential participants based on regional conservation priorities. They often target land buffers around protected areas. Landowners who are awarded contracts receive annual payments if they comply with the contract.

Costa Rica's Environmental Services Payment program is new, but appears to be having some success. On a June 1999 trip, I observed excess demand for conservation contracts among landowners and support for the program from many sectors. However, some issues remain, including minimizing transaction costs, designing and targeting contracts, and developing appropriate institutional rules and roles.

In large part, the design of a direct payment initiative depends on field conditions and conservation objectives. In one region, targeted lands may already be in private hands. In another region, the lands may be publicly owned, but a fraction of the total land will be ceded to local residents, as individuals or as groups. For some ecosystems, a payment for preventing deforestation may be sufficient. In others, bonuses may be paid if periodic surveys indicate the presence of target levels of wildlife. In areas where wildlife are agricultural pests or injurious to humans, compensation payments for damage may also be required (e.g., predator compensation funds of Defenders of Wildlife and the World Wildlife

Fund).

Despite the details that must be addressed, conservation performance payments offer clear advantages over the use of less direct development interventions.

*Simplicity and Appropriate Scales*

With direct payments, practitioners can focus their scarce resources on two key tasks: the design of appropriate institutions and payment schemes. With a smaller set of parameters to influence, practitioners are more likely to achieve their conservation objectives. Furthermore, they can be confident that if a contract is struck, the conservation effect will be positive.

With regard to spatial scale, performance payments are amenable to a landscape approach. For large areas that include different agro-economic zones, the complexity of using development-based interventions to promote habitat conservation is substantial. Practitioners must tailor supporting institutions, infrastructure and appropriate technologies to each zone. Using a contract approach, practitioners need only focus on variations in institutional arrangements across zones. Because performance payments can be targeted more precisely than development interventions, practitioners can also be more confident that their interventions will have an effect on the areas targeted for conservation (e.g., corridors) rather than elsewhere. An OECD study of land diversion programs (OECD 1997:48) noted that in the EU, "[i]mplementation is based on national and regional plans and offers opportunities for flexible targeting and adjustment to local conditions."

Performance payments are also amenable to the short time horizon under which conservation objectives must be met. As soon as the money and the institutions are ready, payments can be made, thus quickly establishing the link between conservation and resident well-being. Practitioners can sustain this link with appropriate financial and institutional design (e.g., endowments). If conditions change dramatically, practitioners can adapt and re-orient by adjusting payment levels, target areas, or institutions.

*Clear Conservation Incentives*

With payments that are conditional on conservation results, the connection between conservation expenditures and objectives is unambiguous to both recipients and donors.

Recipients face a clear choice: protect a parcel of land and receive payments or clear the parcel and forgo the payments. Donors may find conceptualizing and observing the effects of their expenditures easier than with development-based interventions. Funds may therefore be more forthcoming (Simpson & Sedjo 1996).

The explicit connection between payments and conservation objectives also sends a clear signal to residents that ecologically valuable land is economically valuable. In contrast, current conservation efforts often send a signal to residents that they should preemptively clear land lest it be regulated or expropriated. In the Costa Rican payment program, some observers believe that farmers without contracts are forgoing clearing forest in the hope that they may secure a contract in the future (F. Tattenbach, FUNDECOR, per. comm. 1999).

By virtue of the direct link between payments and conservation objectives, performance payments create incentives for local residents to have an *active* stake in protecting ecosystems. In contrast, many indirect development-based approaches (e.g., agricultural intensification) encourage *passive* conservation by local residents. Residents do not make a deliberate choice to protect an ecosystem; the targeted ecosystem is simply not used in productive activities and thus is not degraded. Without active local involvement in conservation, however, many ecosystems will remain open-access resources under continuous threat of conversion.

An increasing amount of evidence indicates that private and common lands are often managed better than government lands for ecological services (Laarman 1995:12). This outcome is especially likely when local institutions can coordinate monitoring and enforcement efforts. Of course an important problem with private control of ecosystems is the divergence between private and social values. With performance payments, however, private agents capture social values attributed to the ecosystem and thus private and social objectives can coincide.

While providing clear benefits to residents, performance payments do not provide large incentives for immigration. Newcomers cannot capture a share of the benefits by simply arriving in a region.

*Rights and Responsibilities*

In the context of performance payments, residents are cast as providers of valuable

services. Their role is changed from adversary to collaborator. This change not only helps to avoid the ethical dilemma of denying poor or indigenous people the ability to earn a livelihood, but it also improves conservation enforcement by creating "citizen guards" who have an active interest in protecting ecosystems. Casting residents as collaborators can also render conservation education more effective. Residents are not told what they are doing wrong, but rather what they are doing right.

At the national and international level, conservation contracts encourage the beneficiaries of ecosystem conservation to pay for those benefits. In particular, the participation of wealthy nations in the conservation of ecosystems in low-income nations has long been recognized as a critical component of global biodiversity protection (Article 20(2) of the Convention on Biological Diversity 1992; World Bank 1992). A controversial aspect of development-based conservation approaches, however, is that much of the financial transfers leave a local area through physical capital purchases and salaries of myriad ex-patriate and host-country experts. Direct payment initiatives ensure that more of the transfers stay in a region.

Host-country governments are also less likely to perceive performance payments as weakening national sovereignty. Industrialized nations are not pressuring low-income nations to set aside lands for protection, but rather they are engaging in a contractual agreement much like any contract for the supply of a service.

## Principal Issues in Conservation Contract Design

Although conservation contracts have advantages over less direct development interventions, they are neither easy to implement nor a one-size-fits-all intervention. Practitioners must address issues of institutional design, property rights, and financing. These issues, however, are also central to the design of development-based interventions. The main advantage of conservation contracts is that practitioners can focus their energies on these issues.

### *Institutional Design and Human Capital Investments*

In order to design a payment program, conservation practitioners must identify the

institutions that will implement the program. Who will raise the money? Who will distribute the money? What institutions will guarantee the rights to benefits distributed by the system? Will coordination among rural residents be required, and if so, how will this be accomplished? How will the legal system be made accessible to rural residents? How will statutory laws and institutions be meshed with traditional ones? To answer these questions, practitioners can learn from direct payment initiatives in the industrialized world and Costa Rica.

Practitioners will also have to face the challenge of designing institutions that can ensure participating rural residents receive their rightful benefits. Institutions must thwart attempts by powerful individuals to divert payments or to use the distribution of benefits as a tool to enhance their power. In so far as conservation contracting adds enforcement eyes to the system, direct payment initiatives may reduce the corruption currently observed in government-controlled natural resource management. Practitioners can learn from recent attempts to use NGO advocates and transparent institutions to share with rural residents the revenues from tourism (Peters 1998) and wildlife culling (Murphree 1993; Muir & Bojö 1994).

A system of direct payments has many of the same institutional requirements as development-based interventions. Both require institutions that can monitor ecosystem health, resolve conflict, coordinate individual behavior, and allocate and enforce rights and responsibilities (Brown & Wycoff-Baird 1992). Unlike more complex development interventions, however, direct payment initiatives allow practitioners to focus their resources on designing the requisite institutions.

*Property Rights*

Closely related to institutional design is the specification of property rights over the contracted areas. Given differences in conservation objectives and in biophysical, cultural, and socio-economic characteristics among regions, there is no single correct way to specify property rights. In some areas, individuals may have, or be given, full, alienable property rights. In other areas, their rights may be more circumscribed. In one situation, rights may be allocated to individuals, while in another case, rights will be allocated to groups. Practitioners can design workable systems by drawing on myriad existing examples of

property rights arrangements. The key task is to ensure that those who invest in conservation have clear, enforceable rights to the benefits from their efforts.

One of the most difficult tasks for practitioners is the identification of the individuals to whom property rights will be allocated. Rights must be allocated to those who can control the use of the resource. The choice of who will, and who will not, receive the rights to payments, and therefore the rights to exclude others from the resource, can produce conflict. Allocating rights so that this conflict does not prevent the achievement of conservation objectives may be one of the most serious challenges to conservation contracting. Of course, brokering the interests of different stakeholders has also been identified as a critical component of development-based conservation projects (Brown & Wycoff-Baird 1992).

In some countries, the rule of law, both traditional and formal, is weak or nonexistent. In such cases, conservation contracting may be impossible. In the same circumstances, however, traditional development interventions or public ownership of ecosystems are also unlikely to lead to desirable conservation outcomes.

*Strategic Behavior and Displacement of Threat*

Practitioners must anticipate strategic behavior by people who will attempt to extract maximum benefits from the program. For example, the promise of payments could encourage people to feign interest in converting lands that would not have been converted in the absence of payments. Residents holding contracts might convert a substitute ecosystem that would not have been converted in the absence of contracted lands.

Residents may also try to exert market power to force conservationists to pay unusually high rents. Practitioners can mitigate the negative consequences of strategic behavior through appropriate institutional design (e.g., U.S. CRP competitive bidding system).

Strategic behavior may also be found in the period prior to project implementation. If there is widespread publicity about conservation payments, practitioners may see an influx of immigrants hoping to be considered “residents” when property rights are allocated. Wealthier and more knowledgeable individuals may engage in land speculation, in the hopes of securing a large portion of the payments. If practitioners anticipate such behavior, however, they can take appropriate actions to mitigate it; for example, practitioners can take

a census prior to negotiations in order to establish baseline conditions.

*Payment Costs*

The notion of paying for people to protect habitat may strike some as an expensive proposition. Many of the regions in which conservation practitioners work, however, are at the margins of the economy where land uses are not very profitable. Analyses of land use around protected areas indicate that residents would accept payments from $28 - $190 per year per hectare to forgo the benefits of ecosystem conversion (Ferraro 1994; Shyamsundar & Kramer 1996; Smith & Mourato 1998). In Costa Rica, annual payments of $35 per hectare generate excess demand for conservation contracts (Calvo & Navarrete 1999).

Practitioners may also find that they do not need to make payments for an entire targeted ecosystem to achieve their objectives. They need only include "just enough" of the ecosystem to make it unlikely, given current economic conditions, infrastructure and enforcement levels, that anyone would convert the remaining area to other uses. Thus, in a well-designed system, not only will residents protect contracted lands near their communities, but they will also protect the remaining ecosystem beyond their lands. The area that constitutes "just enough" may change over time, but with performance payments, practitioners can adjust rights and payments to maintain the required incentives.

The maintenance of biodiversity and other ecological services may also be compatible with some uses (e.g., tourism, extraction of forest products). In these cases, payments would have to compensate residents for a subset of the foregone development options, but not all of them. Unlike indirect investments in eco-enterprise development, however, performance payments achieve conservation objectives regardless of whether or not markets support commercial use of the ecosystem.

The absolute value of performance payments should be evaluated in light of how much money is now being spent on conservation initiatives. Some habitat conservation initiatives have spent up to $1 million per year in very small areas (e.g., Ranomafana National Park Project, Madagascar). Very few, however, have been able to dramatically change local incentives for habitat protection (see for e.g., Wells et al. 1992; Ferraro et al. 1997; World Bank 1997; Oates 1999). Considering the likely costs of using development interventions to create *and maintain* incentives for habitat protection, performance payments

may be very cost-effective over the long run. In *Appendix A*, I show that under plausible conditions, each dollar invested in performance payments yields more conservation benefits than the same dollar invested in indirect interventions.

With a budget equivalent to the U.S. Conservation Reserve Program in 1996 ($1.8 billion), practitioners could make annual payments on up to 60 million hectares. With appropriately targeted payments across the landscape, the actual number of hectares effectively protected could easily be triple or quadruple this amount. To put this area into perspective, consider that in 1996, 309 million hectares were in IUCN protected area classes I-IV in the Middle East,[4] South and Southeast Asia, Central and South America, and Africa (Green & Paine 1997:13).

The performance payments to rural residents would not, however, be the only costs. Practitioners and payment recipients will incur transaction costs in their efforts to design and administer institutions. Although transaction costs in conservation contracting may be significant, many of the same costs are also incurred in development-based interventions. For example, monitoring ecosystem health is a requisite component of both interventions. Through the appropriate design of property rights regimes and institutions, practitioners can minimize these costs.

*Benefits and Risks Associated with the Use of Cash Payments*

Performance payments benefit poor farmers by improving cash flows and providing a fungible store of wealth. For risk-averse farmers, non-stochastic payments also help to diversify the household portfolio and reduce exposure to risk. In the U.S. CRP, risk reduction was an important incentive for enrollment (Gustafson 1994:37).

The same payments, however, can exacerbate residents' exposure to risk by making residents more dependent on markets for meeting their consumption needs. In rural areas, markets are often imperfect, and residents may not be able to transform cash into the resources they need or may be able to do so only at higher prices than anticipated. The same potential problem, however, is prevalent in development-based interventions, particularly commercial initiatives. In contrast to development-based interventions, however, direct payment initiatives do not require households to make significant labor investments and thus permit households to continue production on previously cleared lands or to work off-farm.

Conservation payments can thus be viewed as a *complement* rather than a substitute to current income.

Financial transfers that are conditional upon stopping or limiting what may have been traditional activities can also lead to a variety of social problems. These problems, which are also prevalent in development-based approaches, become more likely the more an activity is associated with the identity of individuals and opportunities to engage in the activity outside of contracted lands shrink.

**Conclusion**

The problem of habitat and biodiversity loss is complex. A complex problem, however, does not always require a complex solution. Conservation practitioners may identify a hundred factors that affect ecosystem use in an area, but they need not design a hundred-pronged intervention to achieve their objectives.

While most of the tropical world continues to experiment with indirect, hydra-headed development interventions to promote ecosystem conservation, some nations are experimenting with more direct contracting approaches that use performance payments to achieve conservation results. Conservation contracting initiatives deserve the attention of practitioners and scholars. Although a contracting approach is neither a magic bullet nor an appropriate intervention for every site, it offers advantages to conservation practitioners in low-income countries:

- it can reduce the complexity of implementation in diverse local conditions;
- it can achieve conservation objectives at the scale of ecosystems, in both the short run and the long run;
- it can permit precise program targeting and rapid adaptation over time;
- it can strengthen the links between individual well-being, individual actions, and habitat conservation, and thus create a local stake in ecosystem protection;
- it can change the role of local residents from adversary to collaborator; and
- it can encourage beneficiaries of ecosystem services to pay for the services.

An International Habitat Reserve Program that facilitates conservation contracting can be an important component of a four-part global conservation strategy: 1) change policies that encourage inefficient habitat conversion; 2) generate livelihood opportunities in regions *far* from threatened ecosystems in order to reduce immigration and encourage emigration away from threatened ecosystems; 3) increase the perceived benefits that local, regional, national, and international citizens receive from natural ecosystems; and 4) design institutions to ensure that those who are in the best position to supply valuable ecological services benefit from their efforts. Given their advantages, conservation performance contracts may be one of the most effective and efficient mechanisms for protecting habitats in low-income nations.

## Appendix A[1]

In this appendix, I use a simple model to illustrate that conservation practitioners can generate greater ecosystem protection under a system of direct payments than under an indirect approach. Consider an "eco-entrepreneur" who produces output, q, under a production function, $f(F, K)$. I will refer to K as capital, but it might be more broadly interpreted as an aggregate of other purchased inputs. I will refer to F as forest, but it can be any ecosystem that the entrepreneur uses in his eco-production activities. Examples of output include tourism experiences, fruits, and eco-certified timber. I assume that competitive markets exist for F, K, and q, with prices $p_F$, $p_K$, and $p_q$, respectively. The price of forest, $p_F$, can be viewed as the opportunity costs of using forest in eco-production (instead of agriculture, for example). Given that markets are assumed competitive, the eco-entrepreneur can be modeled as if he were a profit maximizer (Singh et al. 1986). I also assume that eco-output is positive in the absence of outside interventions, although it may be very low.

Furthermore, I make two assumptions that ensure <u>indirect</u> interventions have a positive impact on the provision of ecosystem services: (1) F and K are technical complements; i.e., $f_{FK} = f_{KF} > 0$, where $f_{FK}$ is the second partial derivative of the production function with respect to F and K; and (2) a unit of forest in eco-production provides the same

quantity and quality of environmental services as a unit of strictly protected (unused) forest. The importance of these assumptions, which are implied in development-based conservation interventions, will be seen momentarily.

Although eco-production is positive in the absence of outside interventions, an outside agent, the "donor," believes that not enough forest is being protected. Given assumption (2) above, the donor's belief is equivalent to believing that the eco-entrepreneur is not using enough forest in eco-production. In other words, intact forest generates positive externalities for the donor and others, but these benefits are not taken into account by the eco-entrepreneur when he makes his production decisions. Thus the donor would like to intervene and encourage more forest protection.

In the model, the donor has two options: *indirect* interventions or *direct* interventions. Indirect interventions render eco-production more profitable by making complementary capital less expensive or by increasing the output price. The underlying assumption of the indirect approach is that such interventions will induce the eco-entrepreneur to use more forest in eco-production, thereby generating more environmental services.[2] For indirect interventions, I consider the subsidization of capital acquisition or the output price. For direct interventions, I consider the donor making a periodic performance payment to the eco-entrepreneur for forest protection (e.g., $X per hectare).

My assumption that a unit of forest in eco-production provides the same quantity and quality of environmental services as a unit of strictly protected (unused) forest has three implications. First, it allows one to compare the indirect and direct interventions without having to consider ecosystem damage that may occur through eco-production activities. The assumption is important because ample evidence suggests that activities like tourism, non-timber forest product extraction, and selective logging can have adverse environmental impacts. Second, the assumption implies that the price of forest, $p_F$, can be viewed as the opportunity costs of conservation, as well as the opportunity costs of using forest in eco-production. Third, the assumption implies that each hectare of forest for which the local agent receives a direct payment can potentially also be used in eco-production. Thus a per

[1] This analysis is based on a more extended analysis that I am conducting with R. David Simpson, of Resources for the Future.

hectare conservation performance payment made by the donor is equivalent to a subsidy on forest used in eco-production.

I begin by comparing a capital subsidy to a forest subsidy (i.e., a direct payment). I assume that the capital subsidy, $dp_K < 0$, and the forest subsidy, $dp_F < 0$, are small and thus I can use calculus to evaluate the local impacts of the subsidies on the production decisions of the eco-entrepreneur. The eco-entrepreneur solves the problem

$$\max_{F,K} \quad p_q f(F,K) - p_K K - p_F F$$

where $f$(F,K) is a continuous, strictly concave, monotonically increasing function. The solution vector (F*, K*) solves the first order conditions

$$(1) \quad p_q f_F - p_F = 0$$
$$(2) \quad p_q f_K - p_K = 0$$

where $f_i$ is the partial derivative of $f$(i, j) with respect to i.

Totally differentiating the first order conditions yields

$$(3) \quad p_q f_{FF}\, dF + p_q f_{FK}\, dK + f_F\, dp_q = dp_F$$
$$(4) \quad p_q f_{KF}\, dF + p_q f_{KK}\, dK + f_K\, dp_q = dp_K$$

Solving this system for dF and dK yields

$$(5) \quad dF = \frac{f_{KK}\, dp_F - f_{FK}\, dp_K + f_{FK}\, f_K\, dp_q - f_F\, f_{KK}\, dp_q}{(f_{FF}\, f_{KK} - f_{FK}\, f_{KF})\, p_q}$$
$$(6) \quad dK = \frac{f_{KF}\, dp_F - f_{FF}\, dp_K + f_{FF}\, f_K\, dp_q - f_F\, f_{KF}\, dp_q}{(-f_{FF}\, f_{KK} + f_{FK}\, f_{KF})\, p_q}$$

[2] I am assuming that increases in capital use simply increase the quantity of output. Quality-improving capital investment, however, is formally equivalent if one assumes the output market is competitive and thus the output price could be considered a function of capital, $a p_q(K)$, where $a$ is an arbitrary constant.

If the donor subsidizes the price of forest by $dp_F < 0$ ($dp_q$, $dp_K = 0$), the use of forest (units protected) changes by

$$(7) \quad dF^D = \frac{f_{KK}\, dp_F}{(f_{FF}\, f_{KK} - f_{FK}\, f_{KF})\, p_q} \quad > 0$$

If the donor subsidizes the price of capital by $dp_K < 0$ ($dp_q$, $dp_F = 0$), the use of forest (units protected) changes by

$$(8) \quad dF^I = \frac{-\, f_{FK}\, dp_K}{(f_{FF}\, f_{KK} - f_{FK}\, f_{KF})\, p_q} \quad > 0$$

Note that equation (8) is positive because I have assumed that capital and forest are technical complements ($f_{FK} > 0$); otherwise, the amount of forest protected would decrease with a subsidy on capital. *In contrast, a conservation performance payment always has a positive effect on forest protection, regardless of the relationship between capital and forest.*

In order to compare the two interventions, I proceed in the following manner. I compare the costs of subsidies on forest and capital that generate the same increase in forest protected. I demonstrate that for a given increase in forest protected, the indirect intervention requires higher capital use. *Thus, in order to generate a given increase in the area of forest protected, society must expend more resources under the indirect intervention. I then show that the donor is also always better off under the indirect intervention (i.e., costs are lower).*

Given a forest subsidy, $\overline{dp_F}$, I solve for a capital subsidy that generates the same increase in the use of forest:

$$(9) \quad \overline{dp_F}\, \frac{\partial F}{\partial p_F} = dp_K \frac{\partial F}{\partial p_K}$$

$$\Rightarrow \quad (10) \quad dp_K = \frac{-\, f_{KK}}{f_{FK}} \overline{dp_F}$$

The left-hand side of (9) is the change in the use of forest under the forest subsidy; the right-hand side is the change in the use of forest under a capital subsidy.

Now I want to know when capital use is greater under the indirect intervention. Let $dK^I$ be the change in capital use under the indirect intervention, and $dK^D$ be the change in capital use under the direct intervention. When is $dK^I \geq dK^D$?

$$\frac{-f_{FF}\ f_{KK}}{(f_{FF}\ f_{KK} - f_{FK}\ f_{KF})\ f_{FK}\ p_q}\overline{dp_F} \geq \frac{-f_{KF}}{(f_{FF}\ f_{KK} - f_{FK}\ f_{KF})\ p_q}\overline{dp_F}$$

$$\Rightarrow f_{FF}\ f_{KK} \geq f_{KF}\ f_{FK} \qquad (10)$$

Given that I assume the production function is concave, equation (10) always holds. Thus, the total social expenditures required to generate a one unit increase in the area of forest protected is higher under the indirect intervention (i.e., $p_F dF^I + p_K dK^I \geq p_F dF^D + p_K dK^D$).

Now, I prove the donor always prefers the direct approach in four steps: (1) I assume the contrary (i.e., there are conditions under which the donor prefers the indirect approach); (2) Given (1), the eco-entrepreneur must prefer the direct approach. If, on the contrary, the eco-entrepreneur also prefers the indirect approach, we would have a contradiction of our social efficiency result above; (3) Given that the direct subsidy is always efficient from a social perspective, the eco-entrepreneur can make a transfer to the donor such that they both prefer the direct subsidy; and (4) I show (3) is not possible and thus (1) cannot hold.

If the donor prefers the indirect approach, the cost of the direct forest subsidy must be greater than the cost of the indirect capital subsidy.

$$(11) \quad (-dp_F\ F^D) - (-dp_K\ K^I) \quad \geq \quad 0$$

where $F^D$ is the optimal level of forest use under the forest subsidy (i.e., $dF^D + F^*$), $K^I$ is the optimal level of capital use under the capital subsidy (i.e., $dK^I + K^*$), and $dp_K < 0$ is the capital subsidy that leads to the use of $F^D$ ($F^*$ and $K^*$ are the original profit-maximizing input levels).

Given that the donor prefers the indirect approach, the eco-entrepreneur must prefer the direct approach. Thus, the change in profit under the direct approach, $d\pi^D$, must be higher than the change in profit under the indirect approach, $d\pi^I$:

$$(12) \quad d\pi^D - d\pi^I \geq 0$$

where

$$(13) \quad d\pi^I = p_q\, dq^I - (p_K + dp_K) dK^I - p_F\, dF^I - dp_K\, K^*$$

$$(14) \quad d\pi^D = p_q\, dq^D - p_K\, dK^D - (p_F + dp_F) dF^D - dp_F\, F^*$$

$$(15) \quad dq^j = f_F\, dF^j + f_K\, dK^j \quad , \quad j = \{I, D\}$$

Given (11) and (12), and knowing that the direct approach is socially efficient, the eco-entrepreneur can theoretically make a transfer to the donor such that they both prefer the direct approach. Thus, the additional profit that the eco-entrepreneur gains when the donor uses the direct approach rather than the indirect approach must be higher than the additional cost incurred by the donor under the same scenario.

$$(16) \quad d\pi^D - d\pi^I \geq (-dp_F\, F^D) - (-dp_K\, K^I)$$

$$\Rightarrow \quad p_q(dq^D - dq^I) - p_K\, dK^D - dp_F\, dF^D - dp_F\, F^* + p_K\, dK^I + dp_K\, dK^I + dp_K\, K^* \geq$$
$$dp_K\, K^* + dp_K\, dK^I - dp_F\, F^* - dp_F\, dF^D$$

$$\Rightarrow \quad p_q(dq^D - dq^I) - p_K\, dK^D + p_K\, dK^I$$

$$\Rightarrow \quad (17) \quad p_q\,(dq^D - dq^I) - p_K\,(dK^D - dK^I) \geq 0$$

(recall that $p_F dF^I = p_F dF^D$ by construction, and thus these terms drop out).

In order to show that the direct approach is socially efficient, I demonstrated that output and capital use is always greater under the indirect approach. Thus the left-hand side of (17) is negative, which is a contradiction. Thus, the donor cannot prefer the indirect approach.

The analytical approach used above can also be used to compare an indirect subsidy on *output* to a direct subsidy on forest. In this case, the results are exactly the same. Given our assumptions of concavity of the production function and profit-maximizing behavior, the direct intervention is preferable to society and the donor.

Although I do not prove it here, it can also be shown that the eco-entrepreneur always prefers the indirect subsidy. Although the welfare of the eco-entrepreneur improves under both approaches, it improves *more* under the indirect approach (i.e., profit is higher). Thus the eco-entrepreneur will have an incentive to encourage the donor to adopt the less efficient indirect approach.

**References**


Angelsen, A. 1999. Agricultural Expansion and Deforestation: modeling the impact of population, market forces and property rights. Journal of Development Economics 58:185-218.

Barbier, E. B. & M. Rauscher. 1995. Policies to Control Tropical Deforestation: trade intervention versus transfers. Pages 260-282 in C. Perrings, K.-G. Mäler and C. Folke, editors. Biodiversity Loss: economic and ecological issues. Cambridge University Press, Cambridge**.**

Barraclough, S. L. and K. B. Ghimire. 1995. Forests and Livelihoods: the social dynamics of deforestation in developing countries. St. Martins Press, New York.

Barrett, C. B. and P. Arcese. 1998. Wildlife Harvest in Integrated Conservation and Development Projects: linking harvest to household demand, agricultural production, and environmental shocks in the Serengeti. Land Economics 74:449-465.

Barry, P.J. 1997. Rates of Return to Public Investment in Agricultural Research and Education. CHOICES (Fourth Quarter):14.

Browder, J. O. 1992. The Limits of Extractivism. Bioscience 42:174-182.

Calvo, A., and G. Navarrete. 1999. El desarrollo del sistema de pago de servicios ambientales en Costa Rica. FONAFIFO and UNDP, San Jose, Costa Rica.

Campbell, B., N. Byron, P. Hobane, E. Madzudzo, F. Matose, and L. Wily. 1999. Moving to Local Control of Woodland Resources – Can CAMPFIRE Go Beyond the Mega-fauna? Society and Natural Resources 12:501-509.

Castro, R., L. Gamez, N. Olson, and F. Tattenbach. 1998. The Costa Rican Experience with Market Instruments to Mitigate Climate Change and Conserve Biodiversity. FUNDECOR, MINAE, and the World Bank, San Jose, Costa Rica.

Conservation International (CI). 1995. Projet de Conservation et de Développement de la Reserve Naturelle Intégrale No. 3 de Zahamena. Plan Annuel de Travail 1995. Antananarivo, Madagascar.

Conservation International (CI), ANGAP, DEF, and KfW. 1996. Projet de Conservation et de Développment Intégré Complexe des Aires Protégées d'Ankarafantsika. Programme de Travail Phase I, 1996-1997 (April 1996). Antananarivo, Madagascar.

Coxhead, I. A. and S. Jayasuriya. 1994. Technical Change in Agriculture and Land

Degradation in Developing Countries: a general equilibrium analysis. Land Economics 70:20-37.
Elahl, A. and B. Khushalani. 1990. Technical Issues of Irrigation Development in Sub-Saharan Africa. Pages 69-82 in S.M. Barghouti and G. Le Moigne, editors. Irrigation in Sub-Saharan Africa: the development of public and private systems. World Bank Washington, D.C.
El Nagheeb, A. H. 1990. Extensification of Agriculture and Deforestation in Sudan: an economic analysis under uncertainty. Ph.D. Dissertation. Department of Agricultural Economics. Madison, WI, University of Wisconsin.
Ferraro, P. J. 1994. Natural resource use in the southeastern rain forests of Madagascar and the local impacts of establishing the Ranomafana National Park. Masters Thesis. Duke University, Durham, North Carolina.
Ferraro, P. J. and R. A. Kramer. 1997. Compensation and Economic Incentives: reducing pressures on protected areas. Pages 187-211 in R. A. Kramer, C. van Schaik and J. Johnson, editors. Last Stand: protected areas and the defense of tropical biodiversity. Oxford University Press, New York.
Ferraro, P. J., and R. Tshombe, R. Mwinyihali, J.A. Hart. 1997. Projets Intégrés de Conservation et de Développement: un cadre pour promouvoir la conservation et la gestion des ressources naturelles. Working Paper no. 6. Wildlife Conservation Society, Bronx, New York
Foster, A. D., J. Behrman, and M. Rosenzweig. 1999. Population Growth, Technical Change, and Forest Degradation. Working Paper. Department of Economics, Brown University, Providence, Rhode Island.
Freese, C.H. 1997. The 'Use It or Lose It' Debate: issues of a conservation paradox. Pages 1-48 in C.H. Freese, editor. Harvesting Wild Species: implications for biodiversity conservation. The Johns Hopkins University Press, Baltimore.
Green, M.J.B., and J. Paine. 1997. State of the World's Protected Areas at the End of the Twentieth Century. Paper presented at IUCN World Commission on Protected Areas Symposium on Protected Areas in the 21st Century: From Islands to Networks Albany, Australia, 24-29th November 1997. World Conservation Monitoring Centre, Cambridge.
Gustafson, C. 1994. Rural Economies. Pages 35-38 in When Conservation Reserve Program contracts expire : the policy options : conference proceedings, February 10-11, 1994 / sponsored by Soil and Water Conservation Society with the support of the Agricultural Research Service, Economic Research Service, Extension Service, Fish and Wildlife Service, Forest Service, Soil Conservation Service, and the U.S. Environmental Protection Agency, Arlington, VA. The Society, Ankeny, Iowa.
Howard, A.F., R.E. Rice, and R.E. Gullison. 1996. Simulated Financial Returns and Selected Environmental Impacts from Four Alternative Silvicultural Prescriptions Applied in the Neotropics: a case study of the Chimanes Forest, Bolivia. Forest Ecology and Management 89:43-57.
Jones, J. R. 1989. Human Settlement of Tropical Colonization in Central America. Pages 43-85 in D. A. Schumann and W. L. Partridge, editors. The Human Ecology of Tropical Land Settlement in Latin America. Westview, Boulder, Colorado.
Kaimowitz, D. and A. Angelsen. 1998. Economic Models of Tropical Deforestation: a review. Center for International Forestry Research, Bogor, Indonesia.
Laarman, J. 1995. Government Policies Affecting Forests in Latin America: An Agenda for

Discussion. Environment Division, Social Sectors and Sustainable Development Department, Inter-American Development Bank, Washington, D.C.
Lawrence, D.C., M. Leighton, and D.R. Peart. 1995. Availability and Extraction of Forest Products in Managed and Primary Forest around a Dayak Village in West Kalimantan, Indonesia. Conservation Biology 9:76-88.
Lewandrowski, J., J. Tobey, and Z. Cook. 1997. The interface between agricultural assistance and the environment: chemical fertilizer consumption and area expansion. Land Economics 73:404-427.
Muir, K. and J. Bojö. 1994. Economic Policy, Wildlife and Land Use in Zimbabwe. Environment Department, World Bank, Washington, D.C.
Murphree, M. W. 1993. Communities as Resource Management Institutions. London, International Institute for Environment and Development.
Oates, J. F. 1999. Myth and Reality in the Rain Forest: how conservation strategies are failing in West Africa. University of California Press, Berkeley, California.
OECD. 1997. The environmental effects of agricultural land diversion schemes. OECD, Paris.
Ozorio de Almeida, A. L. and J. S. Campari. 1995. Sustainable Settlement in the Brazilian Amazon. Oxford University Press, Oxford.
Peres, C. 1999. Tropical Forest Disturbance and Dynamics in Southeastern Asia. Trends in Ecology and Evolution 14:217-219.
Peters, J. W. 1998. Sharing National Park Entrance Fees: forging new partnerships in Madagascar. Society and Natural Resources 11:517-530.
Porter, D., B. Allen, and G. Thompson. 1991. Development in practice: paved with good intentions. Routledge, London.
Projet Parc National Ranomafana (PPNR). 1995. Plan Annuel de Travail du Projet Parc National Ranomafana 1995 (Janvier). Antananarivo, Madagascar.
Redford, K. 1992. The Empty Forest. Bioscience 42:412-22.
Richards, M. 1993. The Potential of Non-Timber Forest Products in Sustainable Natural Forest Management in Amazonia. Commonwealth Forestry Review 72:21-27.
Salafsky, N., B. Cordes, J. Parks, and C. Hochman. 1999. Evaluating Linkages Between Business, the Environment, and Local Communities: final analytical results from the Biodiversity Conservation Network. Biodiversity Support Program, Washington, D.C.
Sedjo, R. A. and R. D. Simpson. 1995. Property Rights, externalities and biodiversity. Pages 79-88 in T.M. Swanson, editor. The Economics and Ecology of Biodiversity Decline: the forces driving global change. Cambridge University Press, Cambridge.
Shyamsundar, P. and R.A. Kramer. 1996. Tropical Forest Protection: An Empirical Analysis of the Costs Borne by Local People. Journal of Environmental Economics and Management 31: 129-144.
Simpson, R. D. and R.A. Sedjo. 1996. Paying for the Conservation of Endangered Ecosystems: a comparison of direct and indirect approaches. Environment and Development Economics **1**:241-257.
Smith, N. J. H. 1996. Effects of Land-Use Systems on the Use and Conservation of Biodiversity. Pages 52-79 in J.P. Srivastava, N. J. H. Smith and D. A. Forno, editors. Biodiversity and Agricultural Intensification: partners for development and conservation. World Bank, Washington, D.C.
Smith, J., S. Mourato, et al. 1998. Willingness to Pay for Environmental Services among

Slash-and-Burn Farmers in the Peruvian Amazon: implications for deforestation and global environmental markets. Selected paper presented at American Agricultural Economics Association Annual Meeting, August 2-5, 1998, Salt Lake City, Utah.
Swanson, T. 1995. The International Regulation of Biodiversity Decline: optimal policy and evolutionary product. Pages 225-259 in C. Perrings, K.-G. Mäler and C. Folke, editors. Biodiversity Loss: economic and ecological issues. Cambridge University Press**,** Cambridge.
Tewari, D.D. and J.Y. Campbell. 1996. Increased Development of Non-timber Forest Products in India: some issues and concerns. Unasylva 187:26-33.
VITA, SAF/FJKM, TFMT, and Clark University. 1995. Projet de Conservation et de Développement Intégrés des Aires Protégées d'Andasibe-Mantadia. Plan Annuel de Travail 1995 (Fevrier). Antananarivo, Madagascar.
Wells, M., and K. Brandon, with Lee Hannah. 1992. People and parks: Linking protected area management with local communities. World Bank, World Wildlife Fund, and U.S. Agency for International Development, Washington, D.C.
Wiersum, K. F. 1986. The Effect of Intensification of Shifting Cultivation in Africa on Stabilizing Land-use and Forest Conservation. Netherlands Journal of Agricultural Science 34:485-488.
World Bank, Operations Evaluation Department. 1988. Rural development : World Bank experience, 1965-1986. World Bank, Washington, D.C.
World Bank, Indonesia and Pacific Islands Country Department. 1997. Investing in Biodiversity: a review of Indonesia's Integrated Conservation and Development Projects. World Bank, Washington, D.C.
World Wildlife Fund (WWF). 1995. Project de Conservation et de Développement Intégrés Andohahela. Programme Annuel de Travail 1995 (Mars). Antananarivo, Madagascar.

## Endnotes

[1] This essay is a revised version of a paper "Global Habitat Protection: limitations of development interventions and the role for an International Habitat Reserve" presented at the conference "Adaptive Collaborative Management for Protected Areas: advancing the potential." September 1998, Ithaca, NY.

[2] The development of hybrid seed corn in the U.S., for example, took in excess of forty years from the initiation of research to the seed's widespread adoption (Barry 1998).

[3] Although the CRP's original objective was to reduce soil erosion, the 1990 Food, Agriculture, Conservation and Trade Act put greater emphasis on improving water quality and wildlife habitat.

[4] The Middle East includes 741,000 square kilometers in two Saudi Arabian protected areas.