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Cost-effective conservation when eco-entrepreneurs have market power

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ABSTRACT. International conservation investments are often made in the form of subsidies to purportedly eco-friendly enterprises rather than as payments conditional on habitat protection. Previous research demonstrated that direct payments for habitat protection are more cost effective than indirect subsidies for the acquisition of complementary inputs used in eco-friendly enterprises. In contrast to this earlier research, we assume in this paper that an 'eco-entrepreneur' may have market power. Market power is shown to compound the advantage of direct payments. Through a simple numerical example, we show that subsidies intended to achieve habitat conservation by encouraging the acquisition of complementary inputs can be spectacularly inefficient. In some cases it would be cheaper simply to buy the land outright. In other plausible cases, the indirect subsidy approach would simply be unable to achieve habitat conservation objectives no matter how much funding were available.

1. Introduction

Governments and citizens throughout the world are concerned with protecting biodiversity through habitat protection. However, many biologically diverse ecosystems, including the majority of tropical rainforests, are located in low-income countries that receive few of the global benefits from their ecosystems. To help and encourage low-income nations to conserve their endangered ecosystems, international conservation and development donors have made substantial investments over the last two decades (estimated at some $10 billion; Ferraro and Simpson, 2003).

To allocate these funds, international donors, host-country governments, and conservation practitioners have experimented with various mechanisms. The most popular vehicle for conservation investment over the last two decades has been the Integrated Conservation and Development Project (ICDP). In the words of one report, ICDPs have 'become the predominant approach to most large-scale internationally financed conservation efforts in developing countries' (CIFOR, 1999). Others characterize ICDPs as 'the now predominant' approach (Van Schaik and Rijksen, 2002), and report that billions of dollars have been invested in ICDPs (Terborgh and Boza, 2002).[1] These initiatives typically provide assistance to ventures that yield commercial outputs and ecosystem protection as joint products. Examples of such eco-friendly ventures include ecotourism, biodiversity prospecting, non-timber forest product extraction, and selective logging. These ventures typically employ relatively undisturbed ecosystems as inputs. The ecosystems are combined with purchased inputs, such as capital and labor, to produce a valuable output, such as tourist excursions, novel chemical compounds, fruits, or timber.

All of the major international development and conservation agencies have made investments to support eco-friendly ventures in endangered ecosystems.[2] Funds are directed towards citizens, communities, non-government agencies, and government agencies that control the fate of ecosystems (these local actors may or may not have fully specified property rights). These funds are used to increase the eco-output price or facilitate the acquisition of complementary inputs, such as tourism infrastructure, product marketing, and processing facilities. The assumption underlying such interventions is simple: local actors, faced with cheaper inputs or higher output prices for an eco-friendly activity, will demand a greater area of intact ecosystem, thereby *indirectly* protecting ecosystems and their constituent services.

An alternative approach to encouraging the conservation of endangered natural ecosystems is to pay for conservation performance *directly*. In this approach, domestic and international actors make payments to individuals or groups that protect ecosystems (Barbier and Rauscher, 1995; Barrett, 1995; Simpson and Sedjo, 1996; Ferraro, 2001; Ferraro and Kiss, 2002). In an earlier article (Ferraro and Simpson, 2002), we demonstrated that, in a perfectly competitive environment, a direct payment approach is a more cost-effective means of motivating local actors to conserve habitat than the more popular indirect subsidy approach. An international donor could achieve a given conservation objective at a lower cost by providing a direct incentive.

The model we used to derive these conclusions assumed perfectly competitive markets in outputs and inputs. Recent work by Albers and Muller (forthcoming) considered how conservation policies (enforcement, aid to improvements in agricultural technology, and direct payments) might be affected by missing input markets. They found that, although

[1] Popular terms for describing similar projects include 'gestion de terroirs' and 'community-based natural resource management'.

[2] Examples include the World Bank, United Nations Environment Program, the Inter-American Development Bank, the Asian Development Bank, the European Union, the bilateral aid organizations of Canada, Germany, the Netherlands, Norway, Sweden, Switzerland, and the United States, and non-governmental organizations such as the World Wildlife Fund, Conservation International, Cultural Survival, and the International Union for the Conservation of Nature (Wells, Brandon, and Hannah, 1992; Brown and Wyckoff-Baird, 1994; Conservation International, 1994; Cultural Survival, 1994; Simpson and Sedjo, 1996; Southgate, 1998; Honey, 1999).

the effectiveness of conservation payments was diminished, the relative effectiveness of payments *vis-à-vis* alternative policies was generally unchanged (the exception being when markets for consumption products were missing; such markets are not modeled explicitly in Ferraro and Simpson (2002) and we return to this context in the conclusions). Furthermore, Albers and Muller find that, when markets were missing, greater enforcement or improvements in agricultural technology could lead to *more* pressure on endangered ecosystems rather than less. Such a perverse outcome was never possible with direct payments.

One might also take exception to the characterization in our previous work of local actors as price-takers who cannot affect the market price of their output. For products extracted from native ecosystems and sold on domestic markets, market power may result from the dispersed nature of market activities and the diminished competitive pressures that exist in the presence of transport costs, which are often substantial in developing nations (Goeschl and Igliori, 2002). Robinson, Williams, and Albers (2002) and Albers (2003) find that the distance over which extractions and marketing takes place is a critical variable in determining the patterns of resource extraction and marketing.

Market power may also exist in the trade of eco-products sold on international markets, such as the $15 billion annual export trade in live animals and plants (WWF/TRAFFIC, 2002). Eco-entrepreneurs involved in the collection and sale of live animals and plants often have market power from licensing restrictions and the restricted ranges of some endemic species (e.g., the tiny golden frog, *Mantella aurantiaca*, which is found only around Perinet, Madagascar). Market power in such markets can also come from product differentiation. Since the late 1980s, conservation and rural development advocates (e.g., cultural survival) have argued that the best way to help rural residents living in forest environments is to develop new products from the wild species under the residents' control.

Furthermore, individuals and communities engaged in non-timber forest product collection and marketing or the provision of ecotourism experiences, to give two examples, often have market power as a result of their location (e.g., gateway towns to an ecosystem) or the unique resources they control. For example, in many areas of the world, unique ecosystems are controlled by private or non-profit organizations (Monteverde cloud forest, Costa Rica), or by communities through community property rights or co-management schemes (in Namibia, for example, communities now have control over large ecosystems and can negotiate contracts with eco-tourism providers). Market power may be particularly important when the 'eco-entrepreneur' is a government agency in charge of large areas of endangered ecosystems. Botswana, for example, has an explicit tourism policy based on restricting the supply of eco-tourism opportunities and charging high prices (Botswana has unusual ecosystems such as the Kalahari Desert and the Okavango Delta; UNDP, 2003).

We demonstrate below that market power considerations make the argument for the cost effectiveness of direct incentives *a fortiori*. The intuition is straightforward. The driving force of the results in our earlier paper was the diminishing returns to the inputs in eco-production. Inasmuch as market power induces further diminishing returns – the gap between marginal revenue and price is greater, the greater is the market power of the seller – indirect mechanisms are still less effective. We demonstrate in numerical examples that not only are indirect subsidies not cost effective, but that they may be worse than simply buying land outright and not conducting *any* activity on it. In fact, one can easily construct examples in which indirect subsidies cannot motivate even relatively modest increases in conservation: the marginal revenue product of additional forestland essentially vanishes.

The next section of the paper introduces the model with which we work and derives the basic cost-effectiveness result with market power. The third section provides a numerical example to make the effects of market power clear. Broader considerations are discussed in the final section.

2. The model

We introduced a stylized model of conservation in the presence of an 'eco-friendly' enterprise in Ferraro and Simpson (2002). In that model, an 'eco-entrepreneur' produces a quantity Q of an eco-friendly product using a production technology, $Q(K, F)$. The production technology represents an economic activity (e.g., tourism) that allows ecosystem services (e.g., biodiversity) to flow relatively unimpeded from the ecosystem used in eco-production. The variable F is referred to as forest, but it can be any ecosystem that the entrepreneur uses in her eco-production activities.[3] The variable K is referred to as capital, but it might be more broadly interpreted as any input or aggregate of other inputs.

The prices of capital and forest are p_K and p_F, respectively (the price of forest, p_F, can be viewed as the opportunity cost of using forest in eco-production instead of, for example, agriculture). We suppose that the eco-entrepreneur may have market power: that is, that the price at which it can sell its output is a function of the quantity of the output it produces. If price is above what would prevail under competition where price equals marginal cost, the degree of market power can be identified (Lau, 1982).

We denote the price of output as p, and state it as a function of production, $p(Q)$, to emphasize potential market power. The eco-entrepreneur's objective is to maximize its profit function

$$\pi(K, F) = p(Q)Q(K, F) - p_K K - p_F F \tag{1}$$

In the absence of outside intervention, the eco-entrepreneur uses, and thus protects, some area of forest for eco-production. However, another actor, the 'donor', wishes to induce greater conservation of forest than the entrepreneur would find profit maximizing under prevailing market

[3] We do not assume that eco-entrepreneurs 'own' the forest, but merely that they can cut it down or use it for eco-friendly activities, and, if they choose the latter, they can receive payments from an outside source for using the forest in an eco-friendly way.

conditions. The donor has two options. First, it can motivate greater conservation *indirectly* by subsidizing the use of capital. Alternatively, the donor can make a *direct* payment for every unit of forest protected.[4]

In our earlier paper we demonstrated that *direct* incentives are more efficient under competitive conditions: the donor can assure the preservation of more forest for the same amount of money, or the same amount of forest for less money, by subsidizing the acquisition of forest land rather than by subsidizing the operations of the eco-friendly enterprise (or by purchasing forest land outright). Of course, donors may also care about the welfare of the communities to which they make donations. If, however, direct incentives are more cost efficient, then direct incentives are superior as both conservation *and development* policy. The cost efficiency of direct incentives implies that a given amount of conservation can be achieved with less money using direct incentives. Thus the difference in the donor's expense under the indirect and direct approaches could simply be given to the eco-entrepreneur, thereby making the entrepreneur better off than under the indirect incentive payment.

The intuition behind the result is embodied in the last observation. There is no compelling argument from either a conservation or a development perspective to acquire capital beyond the point at which the value of its marginal product exceeds its price. It makes more sense for the donor to pay for the things it really cares about: forests and local well-being, than to make payments for capital for the indirect role it plays in achieving both ends. Put in another way, we have a classical Pigovian subsidy argument: forests are providing services beyond their value to local actors, and, hence, the best way to achieve a more desirable level of such services is to subsidize their provision directly.

It is worth asking, then, if market power might reverse our earlier findings. To the contrary, we find that it reinforces them. The relative advantage of direct as opposed to indirect incentives depends on the curvature of the profit relationship. By this we mean that subsidizing inputs will lead to higher profits for the eco-entrepreneur, but that the effects of such subsidies diminish with their size. There are three factors that limit the size of an eco-friendly enterprise. First, there are generally decreasing returns in physical production. Second, there are limitations in the elasticity of substitution between forestland and capital. Finally, and the focus of this paper, there are limitations imposed by the extent of markets. All of these lead to diminishing returns to scale, which, as in our earlier paper, lead to the inferiority of capital subsidies as a conservation strategy.

Decreasing returns to scale in production ought generally be the case when considering eco-production opportunities. Favorable sites for eco-friendly commercial activities are limited by the proximity of transportation networks. 'Natural' sites are, almost by definition, those that are not well served by extensive road systems. Adding more land and more capital

[4] We give the benefit of the doubt to the indirect approach and assume $\partial F/\partial p_K < 0$ and that a unit of forest in eco-production provides the same quantity and quality of environmental services as a unit of strictly protected forest.

will be of little use if one cannot, in effect, change the location of an area (see Robinson, Williams, and Albers (2002) and Albers (2003) for empirical work on this issue in the context of forest extraction activities).

The ability to substitute capital for forestland will also affect the relative effectiveness of indirect conservation payments. If the two are perfectly substitutable, subsidizing capital will have no effect on demand for forestland in the limit. More generally, the less substitutable the two inputs are, the more effective will be indirect subsidies.

The final factor is the elasticity of demand. The less elastic is demand, the less effective will be indirect subsidies: the eco-entrepreneur will have little incentive to increase their use of protected forestland in response to a capital subsidy, if doing so would induce a larger supply of output than the market is prepared to bear.

We might add in concluding this section that we assume an interior solution in the absence of any foreign donations: the eco-enterpreneur would choose strictly positive quantities of forestland and capital, and, by extension, output. It is quite possible that such activities would not be undertaken at all absent a subsidy (Salafsky *et al.*, 1999, find that most of a sample of projects funded by the US government-funded Biodiversity Support Program did not cover their costs). However, if such a corner solution obtained (or no such eco-production function existed), the argument for the superiority of direct incentives is immediate. An unprofitable project is, by definition, one that does not cover its costs, including the opportunity cost of forestland. If a subsidy were required to generate enough variable profits to cover costs, it would be more cost effective to apply the subsidy to the preservation of forestland directly than to the acquisition of complementary capital.

3. Some numerical examples

The above principles are best illustrated by constructing a simple example in which the interacting effects of each can be seen. Suppose that the eco-entrepreneur faces a constant-elasticity of demand function such that $p(Q) = p_0 Q^{-\varepsilon}$.

Suppose that the production function is of the form

$$Q = \left(\beta_K K^{\frac{\eta}{1-\varepsilon}} + \beta_F F^{\frac{\eta}{1-\varepsilon}} + (1 - \beta_K - \beta_F)\, R^{\frac{\eta}{1-\varepsilon}}\right)^{1/\eta} \tag{2}$$

where R is a fixed factor. Recall that K is capital and F forest. We introduce the parameters β_K and β_F to index the relative reliance on these factors in production; other things equal, a larger β_F indicates a greater marginal product of forest in production, and similarly for β_K. This implies that the eco-entrepreneur's profit function is

$$\begin{aligned}\pi &= p_0 Q^{-\varepsilon} \cdot Q - p_F F - p_K K \\ &= p_0\left(\beta_K K^{\frac{\eta}{1-\varepsilon}} + \beta_F F^{\frac{\eta}{1-\varepsilon}} + (1 - \beta_K - \beta_F)\, R^{\frac{\eta}{1-\varepsilon}}\right)^{\frac{1-\varepsilon}{\eta}} - p_F F - p_K K \end{aligned} \tag{3}$$

The parameter ε must be positive and less than one: a monopolist would never operate on the inelastic portion of its demand curve.[5] It will be convenient in what follows to abbreviate $\rho = \eta/(1 - \varepsilon)$. To assure well-behaved solutions, we will assume $\eta < (1 - \varepsilon)$; i.e., $\rho < 1$. Note that the assumption of a monopoly is not the key element driving our results; all that is important is the much less restrictive assumption of market power embodied in a downward-sloping demand curve for the eco-entrepreneur.

The form of expression (3) is familiar: it is the same as that which arises from maximizing profit arising from a constant returns to scale, constant elasticity of substitution production function. Note here, however, that expression (3) will typically *not* reflect constant returns to scale, as we suppose there is a fixed factor, R, that prevents the replication of production possibilities at larger scales. Moreover, the parameter indexing elasticity of substitution, ρ, is now compounded of two expressions. One, η, is related to elasticity of substitution, while the other, ε, measures the quantity flexibility of price (i.e., the inverse of the elasticity of demand).

Deriving the profit-maximizing quantities for K and F is a straightforward exercise, but it is more tedious than enlightening, and we will omit details and simply state

$$K = \left(\frac{1 - \beta_K - \beta_F}{1 - \beta_F \left(\frac{p_F}{p_Q \beta_F} \right)^{\frac{\rho}{\rho - 1}} - \beta_K \left(\frac{p_K}{p_Q \beta_K} \right)^{\frac{\rho}{\rho - 1}}} \right)^{\frac{1}{\rho}} \left(\frac{p_K}{p_Q \beta_K} \right)^{\frac{1}{\rho - 1}} R \tag{4}$$

and

$$F = \left(\frac{1 - \beta_K - \beta_F}{1 - \beta_F \left(\frac{p_F}{p_Q \beta_F} \right)^{\frac{\rho}{\rho - 1}} - \beta_K \left(\frac{p_K}{p_Q \beta_K} \right)^{\frac{\rho}{\rho - 1}}} \right)^{\frac{1}{\rho}} \left(\frac{p_F}{p_Q \beta_F} \right)^{\frac{1}{\rho - 1}} R. \tag{5}$$

Now we can perform a number of exercises to calculate the costs of direct and indirect approaches. In each of the entries in table 1 we have normalized p_0 to ten. We set the fixed factor, R, to one. Other parameters were varied, as indicated at the heads of the columns in Table 1. We then compared direct and indirect approaches as follows. Suppose initially that $p_F = p_K = 1$. From these values we can use (4) and (5) to compute initial values of F and K, and then substitute these values in (3) to compute initial profit.

We then ask what it will cost to induce an x per cent increase in forest area preserved (F), computing costs for $x = 10$, 50, and 100, that is, we ask what it would cost to increase forest area to 1.1, 1.5, and 2.0 the area that would be preserved without the donor's intervention. The overall cost of achieving a given increase in forest area protected consists of two components. The first is the cost of the subsidy required to induce the increase in forest area, whether this cost is in the form of a payment for the use (protection)

[5] It is somewhat problematic to suppose that $0 \leq \varepsilon \leq 1$, as this would imply that the production technology exhibits increasing returns in K, F, and the fixed factor R jointly. By supposing that R is 'large enough', however, we can obviate this concern.

Table 1. *Incremental cost of habitat conservation relative to purchase price of land*

	Parameters			*Percentage increase in forest area desired*		
Scenario	β_k	β_f	ρ	*10%*	*50%*	*100%*
1	0.167	0.500	−3.0	0.093, 0.681	0.373, *	0.572, *
2	0.333	0.333	−3.0	0.095, 0.572	0.378, *	0.578, *
3	0.500	0.167	−3.0	0.104, 0.692	0.402, *	0.599, *
4	0.450	0.450	−3.0	0.085, 0.373	0.351, *	0.550, *
5	0.167	0.500	−1.0	0.037, 0.257	0.187, 4.387	0.319, *
6	0.333	0.333	−1.0	0.047, 0.222	0.200, 1.999	0.333, 2865
7	0.500	0.167	−1.0	0.056, 0.310	0.231, 2.500	0.375, 22.1
8	0.450	0.450	−1.0	0.035, 0.097	0.156, 0.651	0.269, 2.21
9	0.167	0.500	−0.1	0.022, 0.118	0.093, 0.716	0.160, 1.80
10	0.333	0.333	−0.1	0.026, 0.109	0.111, 0.578	0.188, 1.21
11	0.500	0.167	−0.1	0.034, 0.178	0.142, 0.894	0.237, 1.79
12	0.450	0.450	−0.1	0.012, 0.020	0.051, 0.092	0.090, 0.173

Note: *Too large to compute.

of forestland or a subsidy for the acquisition of capital complementary with forestland in the production of the eco-friendly output. The second, offsetting, component of the cost of conservation is the increase in profit afforded to the eco-friendly enterprise.

These two components of cost differ between direct and indirect incentives. Direct incentives are provided by subsidizing the price of *forest*, P_F, until the subsidy is sufficient to induce an $x = 10$ per cent, 50 per cent, or 100 per cent increase in the amount of forest area retained. Indirect incentives are provided by subsidizing the price of *capital*, P_K, until the subsidy is sufficient to induce an $x = 10$ per cent, 50 per cent, or 100 per cent increase in the amount of *forest area* retained. In the final three columns of table 1, we present measures of costs under the dozen different scenarios described in the second through fourth columns. These costs are represented in the following way. Each entry in the last three columns of the table consists of a pair of numbers. The first number in each pair is the ratio of the cost of inducing the indicated increment in forest conserved to the cost of purchasing the increment outright. Paying a subsidy for the acquisition of forestland is necessarily less expensive than would be purchasing an additional hectare of land outright because the eco-entrepreneur realizes a gain in variable profits from the additional hectare used in eco-production.

This upper bound does not necessarily hold for the alternative strategy of conserving additional forestland by subsidizing complementary capital. The second number in each pair of entries in the final columns of the table is the ratio of the cost of inducing the indicated amount of initial conservation to the price of outright purchase. This ratio is not necessarily less than one; it can, in fact, be astronomical. The asterisks in the Table indicate values greater than 10^{99}!

To give an example, in scenario 1, the direct payment approach can achieve a 10 per cent increase in forest area at a cost of 9.3 per cent (0.093) of what it would cost a donor to purchase the additional forest area itself (rather than induce the eco-entrepreneur to acquire the forestland). Under an indirect capital subsidy, the same increase in forest area would costs 68.1 per cent (0.681) of what it would cost a donor to purchase the additional forest area. As described in the previous section, the cost of protection via direct payments is necessarily lower than that of both outright purchase and of protection via 'indirect' subsidies. In the same scenario, the donor could induce a doubling in forest area conserved at a cost equal to 57.1 per cent of the cost of outright purchase. However, not even an enormous capital subsidy could induce the same doubling in forestland conserved.

The variations in costs arise from manipulation of the underlying parameters, β_F, β_K, and ρ. The greater is the sum $\beta_K + \beta_F$, the less important is the fixed factor, R, and thus the less of a constraint it imposes on production possibilities. Hence, cost entries in the 4th, 8th, and 12th scenarios, in which $\beta_K + \beta_F = 9/10$ are lower than in the 1st–3rd, 5th–7th, and 9th–11th scenarios, respectively, in which $\beta_K + \beta_F = 2/3$. Within the three scenarios in which $\beta_K + \beta_F = 2/3$, the advantage of the direct approach is greater the greater the weight placed on forest relative to capital in production (i.e., $\beta_F > \beta_K$).

The 1st–4th, 5th–8th, and 9th–12th scenarios differ in their assumed values of ρ, with $\rho = -3$, -1, and -0.1, respectively. Recall that this parameter is an abbreviation for $\eta/(1 - \varepsilon)$, where ε measures the elasticity of demand, and η may be thought of as a proxy for the elasticity of substitution. By analogy to the case of a constant elasticity of substitution production function, it seems reasonable to confine our attention here to instances in which $\rho < 0$; that is, when the elasticity of substitution between arguments reveals them to be complementary as opposed to substitutable: if inputs are substitutes, a policy of indirect incentives could not be very effective, as the essence of such a policy is to motivate the purchase of complementary inputs.

The table shows that indirect incentives are not very cost effective in any event. Our parameter choices of $\rho = -0.1$, -1.0, and -3.0 demonstrate that indirect incentives are not very cost effective even under relatively generous assumptions, and can be disastrously inefficient under less generous ones. Market power is greatest when the absolute value of ρ is greatest. The greater is the eco-entrepreneur's market power, the less cost effective are all incentives to induce greater forestland use in eco-production (i.e., the cost ratios increase). However, the indirect capital subsidy performs relatively worse in comparison to the direct payment as market power increases (the direct payment's cost ratio becomes a smaller fraction of the indirect subsidy's cost ratio, implying that the indirect subsidy becomes relatively more costly).

The table illustrates that it is not difficult to construct examples in which the costs of protection via outright purchase are less than those of protection via capital subsidies (i.e., scenarios in which the cost ratio is greater than one). In many instances – and here we believe we are making both a conceptual and an empirical statement – conservation donors would be better off forgetting about eco-friendly enterprises *entirely*, and simply buying land for protected areas.[6] As we have noted, the asterisks in the table point to a sobering possibility. For some, not obviously implausible, parameter values and objectives, an indirect approach simply will not achieve the conservation objective at any reasonable cost.

The parameter values we have chosen are for purposes of illustration. It would be extraordinarily difficult to estimate the parameters of an actual profit function for an eco-friendly production activity, and we cannot imagine realistic circumstances under which one could reasonably have much faith in the results of such an exercise. We would, however, suggest that we the model we have chosen for our illustration is relatively flexible. It demonstrates that the cost *dis*advantage of indirect relative to direct conservation incentives varies from modest to astronomical. This reinforces the strong *prima facie* argument for favoring direct over indirect incentives.

4. Conclusion

The analysis above clearly neglects some features of the conservation landscape in developing nations. Markets for intact ecosystems are often absent, or are imperfect in that the costs of enforcing property rights are prohibitive. There are a host of other issues that might also be clumped under the rubric of 'transactions costs'. In this respect, however, a system of conservation performance payments is no worse than indirect interventions. Both require institutions that can monitor ecosystem health, resolve conflict, coordinate individual behavior, and allocate and enforce rights and responsibilities (Ferraro and Kiss, 2002).

It seems clear that no conservation policy will prove effective absent stronger property rights than can now be asserted in much of the developing world. While conservation advocates have often recognized this point, its corollary seems often to have made less of an impression: the certainty with which property rights are established and enforced is a function of the benefits that accrue to ownership. While conflicts may arise when new benefits to ownership are discovered (think, for example, of the violence that plagued colonial mining regions), claims tend to be sorted out relatively quickly. If foreign donors contribute enough to make the establishment of property rights remunerative, the property rights will soon appear and be clarified. Our sense is that much of the appeal of ICDPs is the hope that their establishment will strengthen local people's sense of 'ownership' of their biological resources. From an economic perspective, however, such a hope may confuse cause with effect. Strong rights of ownership result when the benefits of secure tenure are significant (e.g., Barzel, 1997; see also Hotte **Q1**
et al., 2000 and Grossman, 2001, for recent research on the evolution of **Q2**
property rights, and Ferraro and Simpson, forthcoming, for a preliminary **Q3**
discussion of these issues in a conservation policy context). Simply asserting ownership need not necessarily lead to sustainable use.

[6] Of course, an outside agent may not be able to enforce its property rights against local claims even after purchasing the property. This inability to enforce property rights, as well as the ethical issues associated with outside control of local resources in poor rural areas, has lead to the reliance on local incentives to induce conservation performance.

We should note in passing that, while foreign donors might clarify rights of ownership be increasing the benefits of ownership, such 'clarification' could come at the expense of marginalized groups within a society. The poor often have access to endangered ecosystems precisely because the ecosystems are not valuable commercially. *Any* conservation initiative that raises the value of intact ecosystems may increase the demand of outsiders to secure rights to these ecosystems. Without advocates, the poor who depended on the ecosystems for their livelihoods may find themselves substantially worse off after a conservation success story has unfolded.

As mentioned in the introduction, other authors have examined conservation scenarios in which markets were missing (Albers and Muller, forthcoming) or imperfect (Robinson *et al.*, 2002). The same analyses modeled rural residents as utility maximizers with production possibilities and time and resource constraints. With one exception, these models do not contradict the results of our simpler model. The exception is in the case in which there are no markets for consumption goods (which we have not modeled above). In such a scenario, neither the direct payment approach nor the indirect subsidy approach would work, because residents cannot transform cash into food (in such a 'full-belly' context, improvements in agricultural technology may be an appropriate conservation strategy; see Angelsen, 1999). Including a temporal aspect to the eco-entrepreneur's decision likewise does not alter the conclusion that the direct payment approach is more cost effective (in fact, the dynamic analysis suggests that such an approach also leads to higher household income under plausible scenarios; Conrad and Ferraro, 2002).

In this article, we do not explore the reasons why the indirect approach has become the preferred approach in conservation circles. Other authors have considered this question (Simpson and Sedjo, 1996; Ferraro, 2001; Ferraro and Simpson, 2002; Ferraro and Kiss, 2002; Ferraro and Simpson, 2003). We do note, however, that, although the obstacles to implementing a payment approach deserve careful consideration, a direct payment approach to ecosystem protection cannot be dismissed as impractical. Direct incentives are now being put into practice in more than a dozen developing nations (Ferraro and Kiss, 2002; Kiss, forthcoming).[7] Moreover, we do not dispute the wisdom of making *profit-maximizing* investments in eco-friendly commercial activities. Our point is only that, if investments in eco-enterprises are not financially wise, as we suspect is the case in many instances, they will not be cost effective in promoting conservation either.

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Summary

Cost-effective conservation when eco-entrepreneurs have market power

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International investments in ecosystem conservation are often made in the form of subsidies to purportedly eco-friendly enterprises. Examples of such eco-friendly enterprises include ecotourism, biodiversity prospecting, non-timber forest product extraction, and selective logging. These ventures typically employ relatively undisturbed ecosystems as inputs. The ecosystems are combined with purchased inputs, such as capital and labor, to produce a valuable output, such as tourist excursions, novel chemical compounds, fruits, or timber. Conservation funds are directed toward increasing the eco-output price or facilitating the acquisition of complementary inputs, such as tourism infrastructure, product marketing, and processing facilities. The assumption underlying such interventions is simple: 'eco-entrepreneurs' faced with cheaper inputs or higher output prices for an eco-friendly activity will demand a greater area of intact ecosystem, thereby *indirectly* protecting ecosystems and their constituent services.

An alternative approach to encouraging the conservation of endangered natural ecosystems is to pay for conservation performance *directly*. Ferraro and Simpson (2002) demonstrated that, if eco-entrepreneurs engaged in eco-friendly activities act as price-takers, making payments conditional on the area of ecosystem protected is a more cost-effective means of motivating an eco-entrepreneur to conserve habitat than the more popular indirect subsidy approach. A donor could always achieve a given conservation objective at a lower cost by providing a direct incentive.

The essence of many arguments for undertaking eco-friendly enterprises is that eco-entrepreneurs can earn rents from exploiting unusual assets. They have market power. In this paper, in contrast to Ferraro and Simpson (2002), we assume that an eco-entrepreneur may have market power. Market power is shown to compound the advantage of direct payments. Through a simple numerical example, we show that subsidies intended to achieve habitat conservation by encouraging the acquisition of complementary inputs can be spectacularly inefficient. In fact, subsidizing eco-friendly activities may be more costly than simply buying land

outright and not conducting *any* activity on it. Moreover, one can easily construct examples in which indirect subsidies cannot motivate even relatively modest increases in habitat conservation and thus are completely ineffective.