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The Economics of Endangered Species

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Abstract

Because habitat conversion is the greatest threat to species, this article focuses on economic incentives for private land users to protect habitat. Habitat protection policies that fail to account for private incentives often have unintended negative consequences. Private incentives for habitat protection depend on many factors: whether landowners are compensated for the costs of habitat protection, the design of compensation mechanisms, underlying property rights and security of tenure, the structure of conservation contracts, and whether markets can be created that internalize external benefits of habitat protection. In developing countries, agricultural price, credit, tax, and land tenure policies have important effects on the demand for habitat conversion. Private landowners often can provide crucial information about costs of habitat protection and species or habitat values on their property. Policies that encourage self-reporting of this information can greatly improve the cost effectiveness of policies to identify and acquire sites for species protection.

1. INTRODUCTION

The protection of endangered species is of paramount public interest for a variety of reasons. A given species may be of value to food chains that support human food supplies. A species may be integral to ecological systems. Such systems, and biodiversity in general, are invaluable both intrinsically and to human life, with intricate webs of linkages to food, health, recreation, and welfare, both physical and spiritual. Few of these values are priced in markets. Many take the form of existence (non-use) values, the measurement of which occupies a vast literature in economics (Smith 1996).

Because the value of endangered species to society is not captured by private actors who make decisions that crucially affect their fate (and that of natural environments in general), there are externalities present that imply a public interest in molding private incentives to account for the external effects.

The economic literature on externalities associated with endangered species has focused on two main problems (Swanson, 1994). The first is overharvesting. When species are found in an open access regime, extraction rates will be higher than is economically efficient and species may be harvested to extinction. Species have value, but ill-defined property rights and short-run private incentives combine to threaten them. Overharvesting and its policy remedies have been studied extensively in fisheries economics (dating back to Gordon (1954) and including important early work by Clark (1973, 1976), Spence (1975), Clark & Munro (1978), Clark et al. (1979), and Berck (1979)). Economists have also devoted attention to overharvesting of terrestrial species such as big-game animals (e.g., Barbier, et al., 1990). A related literature studies the effects of trade bans on poaching of endangered species (Barbier 2000; Barnes 1996; Bulte et al. 2003; Bulte & van Kooten 1996, 1999; Burton 1999; Fischer 2004; Kremer & Morcom 2000, 2003; Missios 2004).

The other major threat to species---and the focus of this article---is habitat conversion. There is general consensus that habitat conversion is the main source of species loss globally (Raven 1980, Wilson 1988, McNeely et al. 1990, Watson et al. 1995) and that habitat conservation on private lands is at the heart of species preservation in the U.S. and other advanced economies (Innes, et al., 1998). The problem of habitat conversion is particularly acute in tropical forests that are home to a significant share of the world's species and are undergoing rapid conversion. Habitat conversion is fundamentally a land-use problem. Species are threatened because their habitats have a higher private value in agriculture, housing, or some other economic use.

In this article, we focus on two related aspects of habitat conservation: incentives for species preservation on defined private lands (arguably the case most relevant for advanced economies) and incentives for habitat conversion by farmers on the agricultural frontier (arguably the case most relevant to developing countries). We caution the reader at the outset that our focus generally sidesteps important literatures on benefits derived from ecosystem services, biodiversity, and different/marginal species (see Weitzman 1998 in particular, and work on the value of marginal species by Simpson et al. 1996, Rausser & Small 2000, Costello & Ward 2006). We also generally abstract from the political economy of endangered species management, including work on the allocation of budget resources to predominantly "cute" species (Metrick & Weitzman 1996, Ando 1999), and the role of private politics and ecolabeling in the promotion of species conservation, such as in the dolphin-safe tuna campaign (Baron 2006, Innes 2006, Kirchhoff 2000). While these literatures raise central questions on which species should be protected, and to what extent, we focus primarily on the question of how to protect, given an underpinning premise that species' habitat can have significant external value.

2. HABITAT PROTECTION INCENTIVES ON PRIVATE LANDS

At the forefront of the debate on endangered species policy in the United States is the conservation of species and habitat on private land. Roughly 90% of plant and animal species listed as endangered or threatened are on private property (Brown & Shogren 1998), and over half of listed species have 80% or more of their habitat on private land (FWS 1997). Private costs of habitat protections are substantial. In the Pacific Northwest, for example, salmon and steelhead protections limit development and water use, and habitat of the northern spotted owl limits timber harvesting.

Section 9 of the U.S. Endangered Species Act governs species on private lands, proscribing the "taking" of any endangered animal. A taking has been interpreted to include damage to essential habitat (an interpretation upheld by the U.S. Supreme Court in *Babbitt v. Sweet Home Chapter of Communities for a Great Oregon*, 1995). No compensation is required when private land use is restricted to protect habitat, and horror stories abound about government excesses in implementation of Section 9 (Stroup 1995, Lambert & Smith 1994).

From an economic point of view, the important questions raised, in the context of habitat protection policy, concern the incentives that landowners have to manage their property and incentives that the government has to regulate property use in order to mitigate the negative externalities associated with habitat degradation. Landowners make a variety of decisions that affect social welfare, including the investments in private uses for their land, conservation choices that can affect potential habitat values, and provision of information about species or habitat values on their property that can be invaluable in determining its best use. Government regulators make decisions about which properties to regulate for habitat protection and what conservation measures they should require on regulated lands. Private incentives are driven by the provision or absence of compensation, the design of compensation, the design of rights (such as the ability to exclude biological surveyors from one's land and the burden of proof for enforcement of regulation), and the structure of potential conservation contracts that may be offered to landowners. Beyond its presumed interest in promoting social welfare by mitigating externalities, government incentives can be driven by statutory or constitutional requirements for compensation (or the lack of such requirements), costs of compensation, and the design of rights. In what follows, we survey the economics literature on these incentive issues and implications for the design of government policy.

The starting point is the literature on government "takings" of private property. The takings rubric derives from the Fifth Amendment to the U.S. Constitution, which prohibits a government taking of private property without just compensation. When prohibiting the taking of a species, is the government taking private property? Case law appears clear that regulation of species does not qualify as a property taking that compels compensation (Meltz 1994). However, even if not obligatory, compensation may nonetheless be desirable.

Economists' first reaction to the compensation issue is often an appeal to the logic of Coase (1960). Property rights can define whether the landowner must pay the government in order to avoid a taking, or conversely, the government must pay the landowner when it takes the property. Either way, the two parties will bargain with one another to an efficient outcome, so that the question of who compensates whom is only a matter of distribution---who gets what, but not how much economic surplus is obtained overall.

Problems of timing and observability confound this logic. Landowners make decisions before negotiations take place because, for example, potential habitat values are not known at the time of private investment decisions. Alternately, conservation decisions may be unobservable and, hence, not something that can be stipulated in a bargain.

Also confounding Coasian logic are problems of entry and exit that are well known in the law and economics (Shavell 1980) as well as the environmental economics (Collinge & Oates 1982) fields. Bargaining cannot take place with those who have not yet entered an activity or land use; as a result, anticipated rents derived from a given property rights regime create entry incentives that cannot be accounted for in the bargaining process. That is, property rights matter for efficiency. For example, suppose that private investments are subject to the protections of nuisance law, so that others who harm the private activity must pay for the harm done, but that private damages to public resources (such as wildlife) are not subject to nuisance law. Then, private activities that harm a public resource are not taxed for the harm, and the public must instead compensate the private agents not to harm the resource. For a given set of activities (and resources), the logic of Coase implies that efficient nuisance-avoidance outcomes will be negotiated. However, investment decisions---that is, investments in private activities and public resources---are not the subject of bargaining, and they are distorted by the asymmetric property rules: Public resources will be undersupplied and private ones will be oversupplied. [For recent work on "coming to the nuisance" and effects of property rules on investment and location decisions, see Pitchford & Snyder (2003), Innes (2009).]

2.1 Takings

The starting point in the takings literature is Blume et al. (1984), who consider a two-date single parcel model. At time 1, the landowner makes an investment choice I that determines the land's private use value at time 2, $b(I)$. At time 2, the government observes a potential public use of the land---habitat for an endangered species, for example---that is incompatible with the private use. This public-use value, s, is assumed to be either high (with probability a) or low [with probability $(1 - a)$]. In this setting, the government either takes the land completely, which renders the private investment worthless, or leaves the land completely in private hands. When s is high [higher than $b(I)$], the government efficiently takes the land. Hence, the societal return on private investment is $b(I)$ with probability $(1 - a)$ and zero with probability a. As this is precisely the return earned by landowners when they are paid zero compensation for a taking, efficient investment choices are achieved with compensation that is either zero or lump sum (invariant to I). Conversely, compensation tied to the private value $b(I)$ distorts investment choices; the landowner then fails to consider the full societal loss of the investment value in the event of a taking and overinvests.

This stark benchmark motivates substantial subsequent research to identify motives for positive compensation. First is the presence of landowner risk aversion. Absent compensation, the return distribution to the land investment I is very risky: either $b(I)$ with some probability or zero with the complementary probability (when a taking occurs). Compensation reduces the riskiness of this distribution, essentially providing free insurance to the landowner. Blume & Rubinfeld (1984) argue, therefore, that when landowners are risk averse, the government is risk neutral, and there is no insurance market for takings risks, then compensation improves risk sharing and can therefore enhance efficiency. Central to this argument is the absence of an insurance market. There are two possible reasons for the failure of private insurance:

1. Adverse selection: Landowners may have better information about the likelihood of a taking, giving rise to a "market for lemons" in insurance (Akerlof 1970).
2. Political economy: Landowners may be able to lobby successfully for compensation in the event of a taking; if so, one equilibrium outcome is for no landowner to buy (expensive) insurance, and all to successfully lobby the government for compensation.

Farber (1992) and Kaplow (1986) argue that both possible explanations for insurance market failure are not compelling; insurance companies will have information at least as good as landowners will and are likely to have more political power than landowners.

2.2 Preemption

Even absent risk aversion, landowners make decisions that affect the costs of taking land for habitat, the benefits, or both. They can raise costs by preemptively developing their land and lower benefits by altering their management practices to reduce conservation. Theoretical models for regulatory preemption, drawing upon early arguments by Stroup (1997), are developed in Innes (1997, 2000b). Innes (1997) considers a two-period multiparcel model of land development in which, during the first period (*ex ante*), some land (but not all) is optimally developed and, during the second period (*ex post*), the government observes the value of taking land for a public use (such as habitat or dune restoration, as in the case of *Lucas v. South Carolina*, 1992) and decides how much land to take for this purpose. *Ex post*, it is efficient for the government to take the cheapest (undeveloped) land first. Hence, if a taking (of undeveloped land) is uncompensated, landowners have a powerful incentive to preemptively develop their land *ex ante*, thereby significantly reducing their risk of expropriation. Compensation cures this inefficient incentive for excessive development.

Alternately, preemption may take the form of different management practices, such as harvesting timber in shorter rotations to avoid habitat creation or even overt efforts to "shoot, shovel, and shut up." The case of Benjamin Cone is illustrative. To protect habitat of the red-cockaded woodpecker, a bird that makes its home in old-growth pine forests in the southeastern United States, Cone was prohibited from logging 1560 acres of his land in Greensboro, North Carolina, in the early 1990s. His reaction (Sugg 1993) was, "I cannot afford to let those woodpeckers take over the rest of my property. I'm going to a 40-year rotation instead of a 75-80 year rotation."

Landowners can thus affect the probability distribution of the public use value (s) with their choice of conservation effort (Innes 2000b). Implicitly, these effects may include irreversibilities, for example the negation of any positive habitat value when private use investment is sufficiently large. Such effects, as well as information benefits of delayed development in more general dynamic frameworks, yield option values of conservation that motivate less development (Arrow & Fisher 1974). One way to elicit efficient choices of conservation in these cases is to pay Pigovian compensation, namely, pay the landowner the actual public value s whenever the land is taken (Hermalin 1995). The landowner then faces the complete societal distribution of returns: the private value (b) when it is less than the public use value ($b > s$) and the public-use value otherwise ($s > b$).

However, Pigovian compensation is extraordinarily costly to the government. From an efficiency perspective, this cost is important because, in practice, the government must raise tax revenue to meet the cost and taxes are almost always distortionary. Income taxes distort labor/leisure choices, corporate profit taxes distort investment decisions, sales taxes distort choices between taxed and untaxed goods, and so on. Fullerton (1991) estimates that deadweight costs of taxes are between 7 and 25 cents on the dollar. (Measurement of this excess burden is complex, linked to provision of environmental goods, and the subject of a growing literature; see Carbone & Smith (2008) and Bovenberg & Goulder (1996), for recent treatments.)

With costs of compensation, the government will want to elicit efficient conservation choices at minimum possible cost to the Treasury. This can be done with negligence compensation of the type proposed first by Miceli & Segerson (1994). Specifically, let compensation be a fixed positive payment p if and only if the landowner exercises at least a given amount of conservation effort c; otherwise, compensation is zero. Moreover, let the payment p be just sufficient so that, by exercising effort c and receiving payment p in the event of a taking, landowners obtain the same expected payoff as when they face a pure no-compensation rule. Then, landowners are willing to exert the requisite effort c, and the government's cost of eliciting that effort (p) is at a minimum. A final twist is that, because the required payment (p) falls when the target level of conservation c is lowered, it is efficient to set c lower than would be efficient absent any costs of compensation.

Economists have recently uncovered empirical evidence of both forms of regulatory preemption. Margolis et al. (2008) showed that landowners preemptively developed their property because they anticipated their property would be designated as critical habitat for the cactus ferruginous pygmy owl near Tucson, Arizona. Similarly, Zhang (2004) and Lueck & Michael (2003) showed that prospects for habitation by the endangered red cockaded woodpecker accelerated timber harvesting in the southeastern United States.

2.3 Information

2.3.1 Information acquisition

The potential value of a given property as species habitat is often not known until a biological survey has been conducted. This introduces another margin of efficiency: costly information acquisition. Polasky & Doremus (1998) modeled this decision and its implications for the optimal design of (*a*) compensation (Must the landowner pay the government to avoid a taking or vice versa?) and (*b*) the burden of proof (Must the landowner prove that a taking is inefficient in order to avoid the taking, or must the government prove that the taking is efficient?).

Let s be the true value of the public use, known only if a survey is conducted; absent a survey, s has the expected value $E(s)$. Now consider when a taking is not efficient absent information, $E(s) < b$. Then the societal benefit of a survey is any positive net gain from a taking that would not otherwise occur, $s - b > 0$. Moreover, the government obtains these benefits, pays the survey costs, and thus surveys efficiently, when it has both the burden of proof and the compensation obligation. These rules have the added advantage of eliciting landowner cooperation in conducting the survey, which can either lower survey costs or, in some cases, be necessary for performing the survey at all.

Conversely, however, when $E(s) > b$, then the net societal benefit of a survey is any net positive gain from avoiding a taking that would otherwise occur, $b - s > 0$. Here, landowners obtain these benefits, pay survey costs, and survey efficiently, when they have both burden of proof and the compensation obligation.

In Polasky & Doremus (1998), the optimal location of the burden thus depends crucially on the *ceteris paribus*, whether a taking would otherwise occur (implying that the government should have the property right) or would otherwise not occur (implying that the landowner should have the property right).

2.3.2 Information self-reporting

Some information about endangered species is often available to landowners even absent costly survey efforts; they may observe the presence of an endangered creature, for example. When landowners know that their property has become habitat, they have at least three options: (*a*) "shoot, shovel and shut up" (SSS), effectively destroying the habitat; (*b*) ignoring the discovery and proceeding (unmaliciously) as they would have done otherwise, which may or may not destroy habitat; or (*c*) reporting the discovery to the Fish and Wildlife Service and working with them to protect the habitat. Assuming species protection is a desired social objective---in the sense that the value of habitat is greater than the value of a competing use---these choices confront regulators with an enforcement problem.

Related enforcement problems have been studied in the self-reporting literature, beginning with Kaplow & Shavell (1994) and Malik (1993). This literature identifies a number of advantages of regulatory strategies that elicit self-reporting from regulated entities (for a survey, see Innes 2001a). Kaplow & Shavell (1994) and Malik (1993) showed that self-reporting enables regulators to achieve a given probability of government monitoring/inspection (for regulatory violations) at lower cost; the reason is that self-reporters do not need to be monitored. Kaplow & Shavell (1994) further showed that self-reporting economizes on the use of imprisonment, a very costly sanction, because self-reporters can be confronted with an equivalent average sanction that is purely monetary. Innes (1999) showed that self-reporting strategies promote socially advantageous post-violation remediation; in the present context, self-reporting promotes habitat preservation. Innes (2000a) argued that, when violators have heterogeneous probabilities of apprehension, self-reporting can be used to tie sanctions more closely to harm caused (species benefits in the present context); this encourages more efficient *ex ante* measures for harm avoidance. Finally, Innes (2001b) showed that self-reporting can be used to avoid costly avoidance efforts---efforts to avoid apprehension, such as an SSS strategy.

To illustrate the application of some of these arguments to endangered species, suppose the following:

1. Species preservation requires the protection of government regulators and yields a benefit V.
2. A competing private use yields benefit D.
3. An SSS strategy destroys species habitat at zero direct cost to the landowner.
4. The property is species habitat with probability $p\varepsilon(0,1)$, and the landowner knows when the property is habitat.
5. The government's cost of an audit is m, and it audits a property with endogenous probability ρ.
6. The probabilities that an audit discovers SSS and present species are $s < 1$ and $q \geq s$, respectively;
7. The government can levy fines for SSS or an unreported species, respectively, of f_S and f_N, both limited by a maximum sanction $\underline{f}$, and it can compensate self-reporters C.
8. The social cost of funds is $\lambda\varepsilon[0,1)$.
9. Finally, benefits of habitat exceed costs, $V > (1+\lambda)D$.

Assume a species has been discovered by the landowner. Under an SSS strategy, the landowner obtains the payoff $B_{ss} = D - \rho s f_s$. Under a nonreporting strategy, he obtains the payoff $B_{NR} = D(1-\rho q)-\rho q f_N$. And under a self-reporting strategy, he obtains $B_{SR} = C$. Subject to $f_s \leq \underline{f}$, an enforcement regime elicits self-reporting at minimum cost C by satisfying

$$B_{SR} = B_{NR} = B_{ss} \leftrightarrow f_s = \underline{f},\ f_N = (s\underline{f}/q) - D \equiv f,\ C = D(1 - \rho q)-\rho q f. \quad (1)$$

The government's cost of monitoring per landowner is $\rho(1 - p)m$ under self-reporting, namely, the cost of monitoring nonreporting landowners with probability ρ. With nonreporting, this cost is ρm. Welfare under a self-reporting policy that satisfies Equation 1 is thus

$$W_{SR} = p\{V-\lambda C\} - \rho(1 - p)m + (1 - p)D = p\{V - \lambda D + \lambda\rho q(D + f)\}-\rho(1 - p)m +(1 - p)D, \quad (2)$$

namely, the benefit of species protection (V) less social costs of compensation (λC) when a species arrives on the property (probability p), less costs of monitoring, plus benefits of development when a species does not arrive on the property [probability $(1 - p)$]. Similarly, under a nonreporting policy (that satisfies $B_{NR} = B_{ss}$ in Equation 1 but sets $C < B_{NR}$),

$$W_{NR} = p\{D(1 - \rho q) + \rho q(V + \lambda f)\} - \rho m + (1 - p)D. \quad (3)$$

And under an SSS policy (where $B_{NR} \leq B_{ss}$),

$$W_{ss} = D + \lambda\rho s f_s - \rho m. \quad (4)$$

Because the SSS policy is socially destructive, we have (for any given ρ)

$$W_{NR} - W_{ss} = p\rho\{q(V - D) + \lambda(qf - sf_s)\} \geq p\rho q(V - D) > 0, \quad (5)$$

where the inequality follows from the definition of f and $f_s \leq \underline{f}$. Because self-reporting yields both enforcement economies (the Kaplow-Shavell-Malik effect) and certainty in species preservation (the Innes (1999) effect), we also have (for given ρ):

$$W_{SR} - W_{NR} = p(1 - \rho q)(V - (1 + \lambda)D) + \rho p m > 0.$$

In sum, a policy that elicits self-reporting of species at minimum cost---the policy defined in Equation 1---is optimal.[1]

2.4 Voluntary Conservation Agreements

The compensation/takings literature is based on two key premises: (*a*) *Ex ante* (at time 1), landowners make a noncontractible decision that affects the *ex post* (future) distribution of returns to land in private and public uses, and (*b*) *ex post* (time 2), the government makes a 0-1 takings decision, and landowner compensation may be paid. Recent work on voluntary conservation agreements (VCAs) instead assumes that *ex ante* conservation decisions are contractible and regulated with some probability.

Segerson & Miceli (1998) provided the starting point for this new literature. In a single period model, a voluntary landowner conservation choice, c_{v1}, costs $a_v c_{v1}$, and a mandated conservation level, c_{m1}, costs $a_m c_{m1}$, where $a_v \leq a_m$ (there are nonnegative costs of enforcing mandates). Benefits of conservation (gross of costs) are $B_1(c)$. Absent a VCA, regulation occurs with probability p. When regulation occurs, the government maximizes social welfare, c^*_m = argmax $B_1(c)$ - $a_m c$. A VCA can be signed *ex ante*, before it is known whether regulation would otherwise occur; when signed, the VCA displaces any regulation and, in its simplest form, stipulates a conservation level c_{v1} and no compensation or cost sharing.

In this setting, regulation suffers from two inefficiencies: (*a*) Costs of conservation can be higher than with regulation, $a_v < a_m$, and (*b*) regulation can occur with probability less than 1. If neither of these premises hold, so that $a_v/a_m = p = 1$, then regulation is efficient and there is no motive for a VCA. However, if $p < 1$ and/or $a_v < a_m$, then, as Segerson & Miceli (1998) show, a simple (no compensation) VCA will be signed and will enhance efficiency to an extent that generally rises with the relative bargaining power of the regulator. Although proof of this result is somewhat subtle, the intuition is that there are efficiency benefits of locking in a certain (nonstochastic) conservation level---versus the lottery (0, c^*_m)---that enable mutual gains from the VCA. Allowing for compensation/cost-sharing inducements for landowners to sign VCAs, even with a positive social cost of funds, can only increase the scope for a VCA to enhance efficiency.

Langpap & Wu (2004) extended this analysis in some important directions. First, they considered a two-period model with second-period uncertainty about benefits of conservation, $B_2(c_2,w)$, where w is random. Second, they considered irreversibilities in conservation, which they modeled with the constraint $c_2 \leq c_1$. Third, they considered a "no-surprise" clause of a VCA, which guarantees landowners that they will be required to meet a fixed (contractually stipulated) time-2 conservation target, c_{v2}, and no more (or less). Conservation irreversibilities give rise to added costs of regulatory mandates because regulators are assumed to be myopic, ignoring the option value of a higher c_1 in permitting higher levels of c_2. No-surprise clauses add a cost to VCAs because they deny regulators the flexibility to tailor *ex post* (time-2) conservation requirements to *ex post* circumstances (the realization of w); however, they also add the benefit of voluntary (versus mandatory) conservation in time 2, which is advantageous when costs of voluntary conservation are lower ($a_v < a_m$). Absent irreversibility, and with a no-surprises clause, benefits of a VCA (due to $p < 1$ and $a_v < a_m$) must offset their costs in lost time-2 conservation flexibility in order for a VCA to enhance efficiency (and thus be signed).

Because VCAs are not always advantageous, Langpap & Wu (2004) can characterize when they are more likely to arise, implications that Langpap's (2006) empirical work broadly supports. In particular, a VCA is more likely to arise when the probability of regulation is lower (although this also induces a lower level of conservation), when the cost advantage of voluntary (versus mandatory) conservation is larger, and when a no-surprises clause is present. The latter conclusion is unexpected given the assumed cost of no surprises in reduced (time-2) conservation flexibility; it is driven by an assumed form of surprises in Langpap & Wu (2004) that is particularly inefficient.

Two limitations of the analysis by Langpap & Wu (2004) are apparent. First, as Innes et al. (1998) observe, a no-surprises clause effectively requires that the government compensate for departures from the contractual conservation level. Admitting compensation vitiates the cost of a VCA in lost conservation flexibility without sacrificing cost economies of voluntary (versus mandatory) conservation. Second, however, compensation (redistribution from taxed agents to compensated agents) has social costs that need to be taken into account.

Absent social costs of compensation, a VCA with a no-surprises clause is always optimal. A no-surprises clause compels compensation for an *ex post* departure from a prespecified conservation target. (Note that the compensation may be a tax if the *ex post* c_2 is less than the agreed target.) The compensation induces voluntary (versus mandatory) time-2 conservation, with attendant cost savings ($a_v < a_m$). Because the compensation is free, the government can achieve first-best conservation levels using the VCA:

$$c_1^*, \{c_2^*(w)\} = \text{argmax } \{(B_1(c_1)\text{-}a_v c_1) + \int_w (B_2(c_2,w)\text{-}a_v c_2)g(w)dw\}.$$

Although the optimization here ignores irreversibilities, it need not do so; however, the government must of course be forward-looking (not myopic) when designing the VCA in order to account for the positive option value of higher time 1 conservation (Arrow & Fisher 1974). With $p < 1$ and $a_v < a_m$, pure (mandatory) regulation cannot achieve this optimum, even when allowing (plausibly) for non-myopic government decisions.

As the literature on VCAs now stands, the broad conclusions are that VCAs can enhance efficiency; that no-surprise clauses, with compensation for *ex post* departures from preagreed conservation levels, are advantageous; and that the background threat of regulation enhances the ability of a VCA to achieve desired conservation at minimal cost to the government Treasury. Further work is needed to study the optimal design and effects of VCAs when there are positive social costs of compensation (due to deadweight costs of taxation), a nonbenevolent regulator, and potential asymmetric information.

Allowing for a nonbenevolent regulator adds an important dimension to the problem. Amacher & Malik (1996, 1998) studied bargaining between a regulator and a polluter over pollution regulation, where the regulator's objective function is a potential object of supergovernmental choice. An overarching conclusion from this work is that efficiency can be enhanced by endowing the regulator with a proenvironment objective function. In the present context, nonbenevolent (proconservation) regulator preferences affect not only the background threat of regulation---increasing a landowner's incentive to sign a given VCA---but also the structure of bargaining over the design of a VCA. Effects and efficiency properties of nonbenevolent preferences are likely to hinge on the extent of regulator bargaining power and the social cost of funds. For example, if regulators have all the bargaining power (because they are able to make take-it-or-leave-it VCA offers) and the social cost of funds is low, then proenvironment regulator preferences are likely to tilt outcomes too far in favor of species and against landowners.

Asymmetric information becomes important in designing VCAs when there is a positive social cost of funds. Smith & Shogren (2002) studied a specific problem of VCA design under asymmetric information, with the government offering VCA menus that have two parts: a landowner's acreage contribution to habitat (a_i) and an associated government payment (T_i). Landowners are of two types: developers who have a high opportunity cost of habitat acreage and preservers who have a low opportunity cost (because they place a higher private value on preserved habitat). Landowners know their own type, but the government does not.

While Smith & Shogren (2002) studied a version of this problem with economies of agglomeration across landowners, let us consider a simplified environment [patterned after Lewis (1996) and others] in which the government obtains benefits of any given landowner's acreage contribution $B(a)$, but faces two constraints: (*a*) the landowner must be willing to contribute the acreage in return for the corresponding compensation (the participation constraint) and (*b*) among the two (developer and preserver) contracts offered by the government, each landowner type must prefer his/her own contract (incentive compatibility). A problem can arise because, absent information constraints, it is optimal for the developer to be offered a contract with a lower acreage contribution (because his costs of acreage are higher) and a higher payment per acre (in order to elicit participation). This is a problem because the preserver often prefers the developer's small habitat/high payment contract to his own. If the social cost of government funds is zero, this problem can be corrected by simply offering the preserver a higher payment (so that he is "more than willing" to participate). However, because government funds are costly, a second-best solution involves lowering both the developer's acreage contribution and his payment level so as to preserve his participation incentive only. Because the acreage reduction is more beneficial to the developer than to the preserver, this change reduces the preserver payment needed for him to prefer his contract, and this saves government funds. In sum, asymmetric information motivates a lower conservation level/acreage contribution for landowners that have a high cost of conservation.

2.5 Endangered Species Incentives on Private Lands

The broad message from this literature is that society's interest in habitat preservation is well served by government strategies to provide landowners with positive conservation incentives, backed by the threat of sanctions for bad behavior. Positive incentives can include carefully designed compensation programs for habitat conservation and conservation agreements with no-surprise clauses. Also important are incentives for self-reporting of species on private lands, which require rewards for self-reports rather than the economic penalties that characterize present endangered species laws in the United States.

Omitted from this discussion are the constitutional issues raised by governmental (versus landowner) incentives. When the government is not benevolent (the case that motivates the Fifth Amendment restrictions in the United States), what constitutional/supergovernmental restrictions are needed to promote efficient regulatory conduct? We have largely sidestepped this issue here because, at least in the United States, endangered species protections are generally outside the scope of the Constitution. However, we refer the reader to Fischel & Shapiro (1989), Innes (1997), and, more recently, Brennan & Boyd (2006) for subtleties that arise in the design of supergovernmental restraints, including potential economic costs of pure compensation requirements (Innes 1997) and the need to tailor restraints to the nature of government biases (Brennan & Boyd 2006).

3. HABITAT CONVERSION INCENTIVES ON THE AGRICULTURAL FRONTIER

Now consider incentives for habitat protection in developing countries. Figure 1 illustrates the local, national, and global incentives for habitat conversion or preservation.[2] The marginal benefit curve for local residents is MB_L. Nearby forestland can provide fuel, forage, plants and animals for food, traditional medicines, or other goods of local value. Although preserving the land in forests provides a flow of benefits, there is a marginal cost to the local residents of preservation, *MOC*. This is the marginal opportunity cost of not converting the land to pasture or crop production. Given the marginal benefits of land conversion, local inhabitants would choose to preserve land up to H_L.

Figure 1 The local, national, and global incentives for habitat conversion and preservation. From left to right, the figure shows the marginal benefits of preserving more land, such as, a tropical forest. MB_G, MB_N, and MB_L are the marginal benefits of preserving the habitat for the world, for the country where the habitat is located, and for local residents; H_G, H_N, and H_L represent the hectares of habitat the global community, the country where the habitat is located, and local inhabitants, would choose to preserve. *MOC* is the marginal cost to the local residents of preservation. From right to left, the *MOC* curve maps out land conversion's marginal benefits to local inhabitants.

<<Instructions to Composition: insert Figure 1 here>>

Figure 1 reflects a basic incentive problem. The marginal benefits to the country where the habitat is located are MB_N. The unconverted forestland can provide a number of national public goods to the country---such as watersheds, erosion control, or ecotourism revenues---not captured directly by local inhabitants. The government would desire to preserve $H_N > H_L$, but moving there would reduce local welfare by the area *a*. Nationally, the willingness to pay for such a move would be $a + f$. If transactions costs were not too large, the government could improve national welfare by compensating local residents and shifting toward H_N.

Externalities also exist at the international level. Global benefits from preservation would include the expected value of crop germplasm or medicinal plants. Other benefits would be existence values placed on forests or rare species (Brown 1987) or rainforests as carbon sinks to limit global warming (Sandler 1993). Global benefits would also incorporate some measure of option value (Arrow & Fisher 1974; Fisher & Hanemann 1990). Extinction of a species today irreversibly precludes its use at some future date.

The global marginal benefits of preserving the habitat are MB_G, where global benefits are the sum of national and international preservation benefits. The global optimum level of preservation, H_G, is greater than what the national government would be willing to supply. Global gains from increasing preservation from H_L to H_G equal area $a + b + c + d + e + f$, yet this would come at a cost of $a + b + c$ to local inhabitants.

The marginal opportunity cost of habitat preservation, *MOC*, also represents the demand for habitat conversion. Not surprisingly, factors that affect the demand for agricultural land are crucial to explaining tropical deforestation and habitat loss. Opportunity costs are not static. Rather, they can shift to the left (to *MOC′*) in response to market, technological, demographic, or policy changes that increase the demand for agricultural land on the frontier. For example, *MOC′* might represent the marginal opportunity cost of preservation if the government provides subsidies for land clearing, whereas *MOC* represents costs without subsidies.

Policies to protect habitat can be examined in terms of movements along or shifts in the marginal benefit or marginal opportunity cost curves shown in Figure 1. Establishing biological reserves involves designating a particular level of habitat (say, H_N). Local inhabitants may or may not be compensated for loss of access to land and resources. Bioprospecting contracts, ecotourism development, or marketing of ecofriendly forest products attempt to increase local marginal benefits of preservation (MB_L) by creating markets for goods and services compatible with maintaining habitat.

Direct payment programs subsidize local inhabitants directly to protect habitat. In figure 1, local inhabitants have an economic incentive to preserve H_L hectares of habitat. National benefits, however, are maximized at a greater level of preservation, H_N. Increasing preservation from H_L to H_N would increase national benefits by $a + f$, but would impose a net cost of a on local inhabitants. A direct payment scheme could pay local inhabitants a or more in exchange for their increasing preservation from H_L to H_N.

Integrated conservation and development projects (ICDPs) seek to shift up MB_L by developing income-generating activities that rely on habitat protection. They may also attempt to encourage nonagricultural employment or labor-using technical change in agriculture to shift *MOC* to the right. Ferraro & Simpson (2002) refer to this as "conservation by distraction."

3.1 Selecting Biological Reserves

A common policy to preserve biological diversity is to establish biological reserves, areas where habitat conversion is prohibited and economic extraction of biological resources is prohibited or limited. Conservation biologists and environmental nongovernmental organizations have devoted considerable attention to the problem of setting priorities for which sites to preserve (see extensive references in Polasky et al. 2001, Costello & Polasky 2004, Murdoch et al. 2007). Many identify biodiversity hot spots---areas with significant biodiversity, but facing intense conversion pressure. Biological approaches to site selection have focused on maximizing the number and diversity of species protected without considering differences in the opportunity costs across sites. Ando et al. (1998) and Polasky et al. (2001), however, argued that considering differences in opportunity costs greatly enhances the cost effectiveness of habitat protection. They assumed private landowners would be compensated for land placed in preserves. Murdoch et al. (2007) found that (*a*) opportunity costs in different locations can vary by orders of magnitude, (*b*) site biodiversity and opportunity costs may not be highly correlated, and, consequently, (*c*) priority areas selected considering costs can be quite different from those selected ignoring costs. These studies suggest that accounting for differences in opportunity costs can greatly increase biodiversity preserved per dollar spent---an important consideration for government agencies or nongovernmental organizations operating with limited budgets. Costello & Polasky (2004) extended the site-selection problem to a dynamic setting where annual budget constraints imply sites must be chosen sequentially and where habitat conversion risks change over time. They used empirically based numerical simulations to demonstrate how front-loading conservation spending and site selection to earlier periods can significantly improve biodiversity protection.

These site-selection studies, applied to developed countries, assume landowners will be compensated for the opportunity cost of habitat preservation. In developing countries, areas have often been designated as preserves with no such compensation. In Figure 1, this is equivalent to a national government setting the level of habitat at H_N, imposing a cost on local inhabitants of a. In a developing country context, this "fences and fines" approach raises equity concerns and entails significant enforcement costs (Brandon & Wells 1992). The remoteness of many biodiversity-rich areas often makes enforcement of onerous land-use restrictions infeasible.

3.2 Demand for Habitat Conversion

The marginal opportunity cost of habitat preservation, *MOC*, also represents the demand for habitat conversion. A large body of literature has considered household-level incentives to convert tropical habitats for agricultural production (for surveys, see Angelsen & Kaimowitz 1999, Barbier & Burgess 2001).

3.2.1 Agricultural output prices

Several studies found a positive association between agricultural output prices and deforestation/agricultural land expansion (Barbier & Burgess 1996, Panayotou & Sungsuwan 1994, Elnagheeb & Bromley 1994, Binswanger et al. 1987; Deininger & Minten 1999).

3.2.2 Agricultural credit

Many studies also found a positive association between greater access to credit and deforestation (Ozorio de Almeida & Campari 1995, Barbier & Burgess 1996, Pfaff 1997, Deininger & Minten 2002, Binswanger 1991), although Godoy et al. (1997) found a negative association.

3.2.3 Tax incentives

Binswanger (1991), Thiesenhusen (1991), and Schneider (1994) emphasized the role of tax, credit, and land-settlement policies in encouraging land clearing for crop and livestock production in Brazil. Binswanger (1991) considered how various Brazilian tax provisions combined to encourage deforestation in the Brazilian Amazon. These included treatment of agricultural income, land taxes, capital gains and commodity taxes, and tax schemes for large-scale livestock ranches. These tax policies (combined with other market distortions) accelerated the demand for habitat conversion. Furthermore, they especially encouraged land-extensive cattle production, amplifying the adverse environmental effects of deforestation. Such distortions cause a shift in the marginal opportunity cost of preservation curve from *MOC* to *MOC'* (Figure 1). With the distortions, land in habitat is reduced to *H'*. One way to increase local incentives for conservation is simply to remove these distortions.

3.2.4 Wages and employment

Several studies suggested that higher rural wages and employment levels discourage deforestation by making the required labor input more costly (Holden 1993, Ruben et al. 1994, Bluffstone 1995, Barbier & Burgess 1996, Godoy et al. 1997, Pichon 1997, Deininger & Minten 2002). Others, however, noted that increasing ecotourism wages and employment could encourage in-migration, which places pressure on local resources (Wunder 2000, Taylor et al. 2003, Kiss 2004).

3.2.5 Tenurial security

Several studies have documented how tenurial insecurity reduces incentives for soil conservation and intensive cultivation and encourages land clearing in tropical areas (Thiesenhusen 1991, Feder & Onchan 1987, Panayotou & Ashton 1992, Southgate et al. 1991, Pichon 1997). Other studies have considered the general impact of political instability and weak property protection on deforestation (Alston et al. 1999, 2000; Deacon 1994, 1999; Godoy et al. 1997). Here, the main impact is not through increasing demand for agricultural land. Rather, legal insecurity reduces expected returns for sustainable use of forest resources (the MB_L curve shifts downward). In some cases, settlers in forest frontier areas can most readily gain legal title to land by clearing it. Thus, the attempt to secure property rights accelerates habitat conversion (Mendelsohn 1994, Binswanger 1991, Angelsen 1999, Peuker 1992, Schneider 1994, Southgate et al. 1991, Sunderlin & Rodriquez 1996).

3.2.6 Agricultural input prices and technology

The hypothesized and estimated impacts of changing agricultural technologies and input prices are complex (Angelsen & Kaimowitz 1999, Barbier & Burgess 2001). Other inputs may substitute for or complement converted agricultural land, whereas technical change could be either land using or land saving. De Janvry (1981) and Binswanger (1991) noted that policies encouraging labor-saving mechanization in Latin American agriculture limit labor absorption and encourage migration to frontier areas. However, the impacts of other technology and price changes (e.g., seed, irrigation, fertilizers, etc.) are less certain.

3.2.7 Population growth, poverty, and landholding inequality

Extensive literature addresses links between population growth, poverty, and deforestation, the leading mechanism for habitat destruction. There is a long-lived debate between, on one hand, “Malthusians” who conjecture a downward spiral in which population growth depletes natural resources, fueling more population growth and the ultimate collapse of the natural environment (e.g., Ehrlich, et al. 1993) and, on the other hand, "Boserupians" who conjecture that resource scarcity fuels innovation that conserves resources and increases the material services that they deliver (Boserup 1965, Simon 1996). There is, however, general consensus that population decisions and environmental change both affect one another. Population growth fuels exploitation of open access natural resources (Brander & Taylor, 1998), and environmental degradation fuels demand for children in at least two ways: by attenuating the need for children to manage livestock and fetch water and fuelwood (Nerlove 1991; Dasgupta 1995) and by worsening health status (and raising child and adult mortality), increasing the demand for children for income support (Rosenzweig & Stark 1997). Mitigating this "vicious cycle" are incentives for out-migration from degraded rural environments (Amacher, et al. 1998; Chopra & Gulati 1997) and community action to reverse environmental decline. Household production models can generate this sort of vicious cycle, even when accounting for out-migration incentives, but can also predict "Boserupian" responses to degraded environments, all under a premise of open access to environmental resources (Bhattacharya & Innes, 2008).

Foster and Rosenzweig (2003) present a competing view in which forests are privately held and population growth, by fueling demand for forest products, leads to afforestation. Central to their argument is a different property rights regime (private vs. open access) than generally thought to dominate environment-related decisions in developing countries.

Ultimately, the nature of reciprocal effects is an empirical issue. Substantial evidence documents the negative effects of population growth on environmental health in developing countries, both in cross-national studies and within-nation cross-sections (see Panayotou 2000 and Bhattacharya & Innes 2008 for surveys). A smaller literature documents links between resource scarcity and, respectively, fertility (Aggarwal, et al. 2001, Filmer & Pritchett 2002, Loughran & Pritchett 1997) and migration (Amacher, et al. 1998, Chopra & Gulati 1997). Accounting for the joint endogeneity of environmental and population outcomes, Bhattacharya & Innes (2008) report evidence from rural India that supports predictions of the open-access household production model.

Literature and evidence on the poverty-environment nexus is more limited. Forster (1992) and Thiesenhusen (1991) argued that the main causes of tropical deforestation are rural poverty and inequitable land distribution---conditions that drive peasants onto ecologically fragile frontier areas. These studies considered landholding over a broad geographical area, considering the distribution of land between large commercial operations and small-scale producers. Other studies have considered landholding inequality within the class of smallholders in more localized settings. Among this class of traditional, small-scale producers, landholding inequality can make collective action more difficult, discouraging cooperation to conserve resources (Dayton-Johnson 2000). Yet, such inequality may mean that larger landholders within this group (*a*) capture most of the gains from conservation and (*b*) are better able to impose their preferences on others. The net effect of inequality will depend on these countervailing forces. Zapata-Rios et al. (2005) found evidence that inequality accelerates deforestation in Bolivia; Alix-Garcia (2007) found the opposite result in Mexico.

A number of authors have argued that indigenous (and poor) communities often conserve local environmental resources in spite of their poverty, and that, contrary to the conventional view that poverty drives environmental degradation in developing countries (Duraiappah 1998), the rapacious rich are at least equally to blame for over-exploitation of natural resources (Tiffen, et al. 1994, Swinton & Quiroz 2003, Ravnborg 2003). Consistent with this view, Narain, et al.

(2008) recently reported evidence from rural India that both poor and non-poor households may rely heavily on common property natural resource extraction. Dasgupta, et al. (2005) found negative but insignificant effects of initial poverty on deforestation in Cambodia. A vast literature on the "environmental Kuznets curve" (see Caviglia-Harris, et al. 2009 for a review) has found conflicting evidence on effects of average incomes on different environmental outcomes, but does not study environmental effects of income distribution beyond the average. None of this literature distinguishes between the initial income distribution and changes in the income distribution. Nor does it account for the joint determination between environmental degradation or improvement and changes in income for the poor and non-poor. In their attempt to do so with data from rural India, Bhattacharya and Innes (2009) found evidence for negative effects of initial poverty on forestation, and positive effects of increased incomes for all groups (poor and non-poor), while providing further evidence of the dependence of the poor on environmental/forest resources.

Overall, this literature suggests reason for continued policy interest in population-poverty-environment linkages for the protection of species' habitat. Although much more research is called for, there is evidence that improving the economic well-being of the poor with programs that promote improved health and educational outcomes will reduce incentives for population growth and for exploitation of native forests. Similarly, programs that directly improve the local natural environment, both by reducing incentives for population growth and directly improving the economic status of the poor, may help stem environmental decline.

3.3 Bioprospecting and Conservation Incentives

Bioprospecting is the search among living organisms for compounds that have commercial value as active ingredients in pharmaceuticals, pesticides, and other products. Natural products, derived from plants and animals, remain a basic source of many pharmaceuticals. Soejarto & Farnsworth (1989) estimated one quarter of prescription drugs contained some natural products. Natural products remain a major source of drug discovery, either directly or as blueprints for novel chemicals. Newman & Cragg (2007) estimated 63% of the 973 small-molecule drugs approved worldwide from 1981 to June 2006 were based on natural products.

Whereas many biologists and environmentalists have seen bioprospecting agreements as avenues to improve incentives for habitat conservation, economists generally have been more skeptical. Simpson et al. (1996) argued that, although biodiversity is valuable overall what matters for bioprospecting is the value of a marginal species. They argued that the marginal value of habitat would be low ($21/hectare). When several species produce the same chemical compound, the probability of discovering the compound's value is high, but discovery in one species will render other species redundant as a source of that compound. In cases where a compound is rarely found, the probability of finding a useful lead will be quite small.

Rausser & Small (2000), in contrast, found that marginal values of habitat for bioprospecting could be large (more than $9000/hectare). In such cases, private bioprospecting contracts could create incentives to conserve biological diversity. Rausser & Small (2000) attributed this difference to the role of information search process. Simpson et al. (1996) assumed a random-search process. Rausser & Small (2000) assumed prospectors can use scientific information to search more efficiently, raising the marginal value of new searches.

Costello & Ward (2006) examined the role of information and search processes on marginal values of biodiversity-rich habitats, explicitly comparing the models and results of Simpson et al. (1996) and Rausser & Small (2000). Using the numerical values assumed in both studies, they calculated the marginal value of land in biodiversity hot spots for both random searches (as in Simpson et al. 1996) and optimal searches (as in Rausser & Small 2000). They found use of information in the search process did raise marginal values, but the increase accounted for only 4% of the difference in the results of the two studies. The bulk of the difference came from varying assumptions about other parameter values used in the models. Costello & Ward (2006) then derived ranges of estimates of the marginal value of habitat using ranges of parameter values from existing literature. Based on this exercise, their results support the assertion by Simpson et al. (1996) that the marginal value of land under bioprospecting would be low and insufficient to counter conversion incentives.

Other studies focused on different aspects of bioprospecting problems, but they reached similar conclusions. Barrett & Lybbert (2000) emphasized the difficulties of transferring bioprospecting gains to the poor in tropical countries who are making land-clearing decisions. Even if bioprospecting can increase marginal benefits at the national level (a shift out in MB_N in Figure 1), incentives may not change at the local level (no change in MB_L). Frisvold & Condon (1994), rather than focus on the absolute value of bioprospecting gains, argued that opportunity costs of preservation are large relative to bioprospecting gains and that costs are growing significantly over time. In terms of Figure 1, the *MOC* curve is steep and shifting to the left rapidly, so internalizing external benefits of habitat may have little effect on conservation. Frisvold & Day-Rubenstein (2008) presented an *ex post* study of the anticancer drug taxol, derived from yew tree species. They illustrated how bioprospecting can exchange one extinction threat (habitat conversion, because a species is not valued) for another (overexploitation, because the resource is harvested under open access). The case of taxol illustrates that creating market demand for genetic resources without clearly defining property rights over them can lead to resource depletion rather than conservation. To date, 64 plant species have been listed as threatened under Convention on International Trade in Endangered Species of Wild Fauna and Flora expressly because of the threat of overharvest for medicinal uses (Schippman 2001).

3.4 Integrated Conservation and Development Projects, Ecotourism, and Green Products

ICDPs are usually directed at populations adjacent to biological reserves, parks, or other protected areas in developing countries. These projects aim to develop local income-generating activities that (*a*) are based on marketing goods or services compatible with sustainable use of resources in or near protected areas or (*b*) provide an alternative to habitat conversion (Brandon & Wells 1992). ICDPs, thus, try to combine conservation with antipoverty and rural development goals. Designing a single program to achieve multiple objectives well is difficult. For example, providing employment opportunities near protected areas might reduce poverty, but it may also draw migrants to the area. Increasing local incomes need not discourage land conversion. Brandon & Wells (1992) pointed out that there is little support for assumptions that poor households will cease economic activity (such as land clearing) once an income target is reached. Local residents may actually invest additional income into expansion of agricultural activities (Kiss 2004)

ICDPs frequently have components to encourage ecotourism. The U.S. Agency for International Development, the World Bank, and conservation nongovernmental organizations have supported numerous community-based ecotourism projects (Kiss 2004). Few of these projects have undergone rigorous study or evaluation (Taylor et al. 2003, Kiss 2004). Kiss (2004) concluded that, in remote, biodiversity-rich areas, tourism revenues are seldom large enough to shift land use patterns significantly. However, they may play a role in protecting small, critical habitats such as migration corridors.

Ecotourism can have unanticipated negative consequences. Tourist demand for food can increase local incentives to convert forests to agriculture (Wunder 2000). Although, as noted above, higher wages have been associated with less deforestation, Taylor et al. (2003) suggested that higher wages have encouraged in-migration to the Galapagos Islands. They call this a "tourism-income-population growth spiral" that can place burdens on local resources.

There have also been attempts to develop markets for green products such as shade-grown coffee or tagua nuts (Ferraro et al. 2005). Again, the goal is to shift up the local marginal benefit curve of habitat preservation, MB_L. The idea is to bundle a private good (e.g., coffee) with a public good (habitat conservation). Cornes & Sandler (1996) referred to this as an "impure" public good. Marketing of green products often includes some sort of third-party certification of the environmental benefits created by production or sale of products. An important policy question is whether such certification programs do in fact enhance conservation (Fischer et al. 2005). As with studies of bioprospecting and ecotourism, economists have questioned the scope for green-product marketing to increase marginal benefits of habitat protection significantly (Stevens et al. 1998, Swallow & Sedjo 2000, Hardner & Rice 2002, Sedjo & Swallow 2002, Fischer et al. 2005).

3.5 Direct Payments for Conservation

ICDPs, bioprospecting, and ecotourism/green markets all aim to shift up the local benefit curve from habitat preservation (MB_L in Figure 1). Some studies suggest that, rather than follow this indirect approach, paying for habitat preservation directly may be preferable (Simpson & Sedjo 1996, Ferraro & Simpson 2002, Ferraro & Kiss 2002, Ferraro et al. 2005). In Figure 1, this amounts to paying local inhabitants at least $a + b + c$ to increase preservation to H_G rather than institute an indirect program to shift up MB_L. Empirically based numerical simulations suggest direct payments can be significantly more cost effective (Ferraro & Simpson 2002, Ferraro et al. 2005). Yet, implementation of direct payments in developing countries has been limited, and there is a lack of rigorous *ex post* evaluation of such programs. Research on design and implementation of VCAs in developed countries (discussed above) may provide insights to further study in developing countries.

4. CONCLUSIONS

Habitat conversion is the major cause of species endangerment. In advanced economies, private landowners are often the ones making immediate decisions that affect habitat, including outright conversion of native environments to agricultural or other development uses, investments that enhance or detract from habitat values, or providing information about species or habitat values on their property. In developing economies, incentives to private landowners may also be important, particularly in the establishment of biological reserves. However, in addition, vital to habitat preservation or conversion are incentives faced by local residents to expand the agricultural frontier into primitive forestlands. In this article, we review literature and arguments related to the incentives facing private actors in these two stereotypic situations.

In examining incentives to private landowners for habitat conservation, related literature identifies a number of negative effects of uncompensated government mandates, including creating incentives for early land development that destroys potential habitat, noxious “shoot, shovel and shut up” behavior that kills individual endangered species, and deterring valuable information collection and reporting about species’ status and potential habitat values on private lands. From the policy perspective, a lesson from this work is that positive incentives (carrots) in tandem with a background threat of regulation (sticks) can have salutary effects on the public’s overall species conservation objectives by providing rewards to self-reporting of species-related information, compensating landowners for regulatory or complete "takings" of their property in order to protect species, and a careful design of no-surprises voluntary conservation agreements (VCA's).

Incentive issues on the agricultural frontier are complex, encompassing many macro-economic linkages. For example, population growth, poverty, and the design of property rights are at the heart of incentives for environmental exploitation by rural residents of poor countries. The literature broadly suggests multi-directional linkages between the "environment," population growth, and poverty, with general (but not completely uncontroverted) evidence that population growth and poverty tend to increase deforestation that destroys species habitat. These conclusions suggest the importance, for example, of health and education policy that improves infant and adult health outcomes and educational achievement, thereby alleviating poverty and reducing incentives for population growth, both of which can reduce incentives for forest exploitation on the frontier. They also suggest the importance of direct poverty alleviation programs, and ICDP's designed to protect local environments in tandem with vesting the local poor with an economic stake in preservation of these environments using such measures as entitlements to some non-timber forest products and rents from eco-tourism. The latter policies target the opposite direction of causation in the "vicious cycle" of degradation, with degradation itself fueling proximate causes of further degradation, including population growth and poverty.

Often missing from academic analyses of these issues is a distinction between different types of environmental / forest resources. In particular, private forests cultivated for timber sales often are ecologically simplified plantations that offer only limited benefits in terms of biodiversity. In contrast, native forests may offer rich rewards in species diversity and habitat. Such a distinction is potentially crucial in the design of policy to promote ecological wealth. For example, Foster and Rosenzweig (2003) stress the benefits of population growth in spurring demand for marketed forest products that drive increased supplies of private forestlands. While their logic hinges on well-defined property rights (as opposed to open or near-open access resources) that often are lacking in the rural environments of developing economies (Bhattacharya and Innes, 2008), it also ignores differential ecological benefits of private vs. native forests.

Extant literature suggests limits on the scope for bioprospecting agreements to spur species / habitat conservation, in part due to the ex-post erosion of rents from over-exploitation of a valued but essentially open access species (Frisvold & Day-Rubinstein 2008). And strengthening of property rights need not always help in habitat protection; rather, protecting biologically sensitive or valuable lands by denying property rights to those that clear them (vs. allowing land clearing to be the road to title) and penalizing those that clear or poach on these lands can promote protection in principle. A key component of such a strategy, however, must be local support for conservation, as local communities are essential to enforcement of sanctions, and local support requires vesting local communities with an economic stake in conservation (with ICDP's, for example).

Finally, we have said little about incentives for public sector action to protect species and habitat. Such incentives are front and center to preservation of the world's native tropical forests that house vast treasures of biological wealth and reside principally in countries such as Brazil and Indonesia that have at times promoted forest-clearing policies at complete odds with global interests in biodiversity and carbon sink conservation. This important subject relates to a variety of key literatures, including those on environmental federalism (Oates and Schwabb, 1988; Wellisch, 1995; Kunce and Shogren, 2005), the design of constitutional restraints (Fischel and

Shapiro, 1989; Innes, 1997; Brennan and Boyd, 2006), the design of international environmental agreements (Barrett, 2003) and the political economy of environmental protection (Damania, Fredrisksson and List, 2003). Applying (and integrating) this literature to biodiversity protection is an important challenge for future research.

Summary Points

1. The economic literature on endangered species focuses on two main threats: overharvesting and habitat conversion. The consensus is that habitat conversion is the main source of species loss.
2. Habitat conversion is fundamentally a land-use problem. Species are threatened because their habitats have a higher private value in another economic use. Landowner decisions that affect social welfare include investments in private land uses, conservation choices that affect habitat values, and providing information about habitat values on their property.
3. A broad message from economics literature is that society's interest in habitat preservation is well served by government strategies to provide landowners with positive conservation incentives, backed by the threat of sanctions for bad behavior. Positive incentives for habitat conversion include compensation and conservation agreements with no-surprise clauses.
4. Also important are policies that reward self-reporting of species on private lands rather than the economic penalties that characterize present endangered species laws in the United States.
5. Policies to protect habitat in developing countries include creation of markets for goods and services compatible with maintaining habitat, direct payments to protect habitat, and encouraging nonagricultural employment or labor-using technical change in agriculture to reduce demand for land conversion.
6. Major factors stimulating demand for agricultural land and habitat conversion in developing countries include agricultural output prices; agricultural credit; tax policies; rural wages; and employment, security of tenure, and landholding inequality.
7. Policies to create of markets for goods and services compatible with protecting habitat, such as bioprospecting, ecotourism, or green product promotion, have not received sufficient, rigorous, *ex post* evaluation to determine if they do in fact provide significant conservation incentives.

Future Issues

1. How can voluntary conservation agreements provide better incentives to protect habitat? Addressing this question requires both *ex ante*, normative analysis of how agreements should be developed and *ex post*, positive analysis of the effectiveness of implemented agreements.

2. Do projects encouraging ecotourism, bioprospecting, and green product marketing provide significant incentives for habitat protection? Projects require more rigorous, quantitative evaluation and comparison with the opportunity costs of conservation.
3. What role can agricultural policy play in reducing demand for habitat conversion in developing countries?

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[1]Linearity of W_{SR} in ρ implies a corner solution for the monitoring rate. However, if costs per audit exhibit diminishing returns in ρ ($m = m(\rho)$, where $m' > 0$), then there can be an interior solution.

[2]This graphical approach could easily apply to freshwater habitats where diversions of instream flows for agriculture or hydropower could threaten aquatic species.