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RESOURCES FOR THE FUTURE

Potential Cost-Effectiveness of Incentive Payment Programs for Biological Conservation

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Abstract

This study assesses the potential cost-effectiveness of incentive payment programs relative to traditional top-down regulatory programs for biological conservation. We develop site-level estimates of the opportunity cost and the nonmonetized biological benefits of protecting biodiversity hotspots in Finnish nonindustrial private forests. We then use these estimates to compare and contrast the cost-effectiveness of alternative conservation programs. Our results suggest that incentive payment programs, which tacitly capitalize on landowners' private knowledge about the opportunity costs of conservation, may be considerably more cost-effective than traditional top-down regulatory programs.

Key Words: biodiversity conservation, incentive payments, cost-effectiveness, opportunity cost, biological benefits, non-industrial private forests

JEL Classification Numbers: C42, C46, Q15, Q20, Q21, Q23, Q57

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I. Introduction

Incentive payment programs (IPPs) are used for biological conservation on private lands. These programs address a fundamental problem, the lack of economic motivation for private landowners to provide public benefits through biological conservation (Brown and Shogren 1998), and increasing IPP use is broadly supported (OECD 2003). Spending on existing IPPs is already considerable; the current spending on agricultural land programs alone is billions of dollars (Berstein et al. 2004).

Despite broad support for IPPs, their effectiveness for biological conservation is not well understood and sometimes questioned (e.g., Kleijn et al. 2001; OECD 2003). An important feature of IPPs is that since they are voluntary, their conservation outcomes are products of landowner decisions. First, the conservation agency designs and offers an IPP to the landowners; then, landowners decide whether to participate and, if so, in which areas to enroll. Thus the conservation agency influences but does not completely control program outcomes.

Biological conservation problems are often examined using reserve site selection models (Ando et al. 1998; Arthur et al. 2004; Costello and Polasky 2004; Margules and Pressey 2000; McDonnell et al. 2002; Onal and Briers 2002; Polasky et al. 2005; Williams et al. 2004). These models entail mathematical optimization and the selection of protected areas from all candidate areas by using a centralized decision process. This approach provides a useful benchmark—the most favorable centralized program configuration—but does not address essential questions regarding the quality of IPP conservation outcomes, which are determined by the decentralized enrollment decisions of individual landowners.[i]

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In this paper, we assess IPPs relative to traditional centralized, top-down approaches to biological conservation. We do this by comparing the cost and conservation outcomes of an IPP and a traditional top-down program applied to the protection of small-scale biodiversity hotspots in Finland. Such an assessment requires site-level information about potential biological benefits (to measure the outputs of different conservation programs) and the opportunity cost of conservation at potential candidate sites (to reflect landowners' enrollment in different IPP configurations). Together, estimates of the biological benefits and the opportunity costs of conservation enable the assessment of cost-effectiveness for alternative programs.

Our focus in this paper is to compare the cost-efficiency of alternative approaches to biological conservation. Therefore, we do not monetize the benefits from conservation but examine how cost-efficiently alternative programs reach different biological targets. The monetization of biological benefits is of course of great interest to economists, but site-level monetization for a national program would be challenging, to say the least.

We estimated the opportunity costs of conservation from a national survey of landowners in Finland, who were asked about their willingness to protect specific small-scale habitats that we call "biodiversity hotspots." We combined landowners' biological assessments of their forests with data on primary habitats of endangered species to derive site-level estimates of the biological benefits derived from enrolling a site in the IPP or from setting aside the parcel via a top-down mechanism. We then used the site-level estimates of the biological benefits and the opportunity costs of conservation to examine the cost-effectiveness of achieving different conservation targets by alternative programs. To our knowledge, this assessment is the first to contrast an IPP with top-down regulatory programs by using estimates of the site-level biological benefits and the opportunity cost of conservation elicited directly from landowners.

Biodiversity hotspots typically are small areas—often a hectare or two—but they are among the primary conservation targets in Finland because they provide habitat to nearly a thousand endangered species (including 38 vertebrates, 443 invertebrates, and 11 vascular plants [Rassi et al. 2001]) that are jeopardized by forest and wetland management (Hanski 2005; Hanski and Hammond 1995). The endangerment of forest species is associated with forest management practices that have changed forest composition, caused habitat loss and fragmentation, and decreased the number of decaying trees in the forests (Hanski and Hammond 1995). The Finnish conservation landscape features hundreds of thousands of households that own and manage parcels that provide owners with income while also supporting biodiversity. As such, it represents conservation problems in boreal areas well (Larsson and Danell 2001; UNECE/FAO 2000). However, it also is broadly representative of many other conservation problems throughout the world that are related to biodiversity protection in working landscapes and on private land.

The remainder of this paper is organized as follows. In Section 2, we explain the policy context and comparisons examined in this study. In Section 3, we describe how the opportunity cost and the biological benefits of conservation were estimated, including the design and implementation of the landowner survey, the estimation and use of enrollment in the proposed IPP for predicting the opportunity cost of conservation, and the estimation of the biological benefits of conservation from our survey data combined with data on the primary habitats of endangered species. In Section 4, we describe the prioritization of candidate sites under alternative policy programs. In Section 5, we explain the relative cost-effectiveness of IPPs and top-down regulatory programs. The paper concludes with a discussion in Section 6.

II. Policy Context and Comparisons

Finland has more than 400,000 privately owned nonindustrial forest holdings that are, on average, approximately 37 hectares in size (Finnish Forest Research Institute 2004). Forest holdings are generally comprised of multiple forest stands, each of which typically covers a relatively small area, often not more than a few hectares and usually fewer than 10 hectares. Most forests are managed for commercial timber production, and the management of a forest stand typically follows a 60- to 120-year rotation cycle, depending on geographical location, soil conditions, and tree species.

Beginning in 1997, Finland implemented an IPP for landowners to promote the protection of old-growth forests and biodiversity hotspots. Currently, a landowner who enrolls qualified habitat in the program for 10 to 30 years is eligible to receive an incentive payment, a cost-sharing compensation that is determined by the surface area and timber stock of the protected forest. By enrolling in the program, the landowner is precluded from harvesting or managing the enrolled forest, although the program allows for nondisruptive uses such as hiking and wildlife viewing. A participating landowner receives the incentive payment in full at the beginning of the protection period.

The program has only a few participants to date, but this basic approach will nevertheless play a central role in Finland's plans to expand its protection of privately owned nonindustrial forests (Ministry of Agriculture and Forestry 2001). Finland has already placed under protection contiguous areas that make up more than 10% of its total land area; these areas are mostly in the north, where state ownership is more common and land values are lower. The current program proposes to supplement existing reserves by establishing a network of many smaller protected forest habitats that are dispersed in a mosaic pattern throughout the country.

We evaluate the potential effectiveness of the proposed IPP by contrasting three programs:

- an IPP in which landowners participate voluntarily if the incentive payment exceeds the opportunity cost of protecting a certain site,
- a species-only program in which each site is selected according to the number of endangered species it supports, and
- a cost–benefit benchmark, which identifies the set of sites that would achieve different biological targets most cost-efficiently.

The species-only program is a traditional top-down regulatory approach and thus is a useful benchmark (e.g., Finnish regulations identify protected areas by biological type; in the United States, critical habitat designations under the Endangered Species Act are determined primarily by species presence). The cost–benefit benchmark provides another helpful yardstick—the least costly design to reach different biological conservation targets—but is difficult to use in real policy applications because it requires extensive site-level information on biological benefits and the costs of conservation.[ii]

III. Estimation of Opportunity Cost and Biological Benefits of Conservation

Survey of Nonindustrial Private Forest Owners

We conducted a survey of nonindustrial private forest owners, randomly selected from all such landowners in Finland. This population owns and manages most of the currently unprotected forestland in the country (Finnish Forest Research Institute 2004). The survey was sent to 2,400 private nonindustrial forest owners randomly identified from the National Bureau of Taxation database, which includes owners of every hectare of Finnish forestland.[iii] Of 2,380 surveys that reached the intended respondents,[iv] 1,129 responses (47.4%) were returned—a rate similar to or higher than that of surveys conducted to estimate the public's willingness to pay for different conservation programs in Finland (Li et al. 2004, Siikamäki and Layton forthcoming).

The survey questionnaire was designed in consultation with ecologists, nonmarket survey experts, foresters, forest and farm survey experts, and forest owners. Preliminary testing included face-to-face interviews and a pilot survey of 200 forest owners. The cover letter indicated that the survey responses were strictly confidential and asked that the survey be completed by the person in charge of forest management decisions. The questionnaire—which included a variety of questions about the landowner, the landowner's forests and their management, and the types of biodiversity hotspots in those forests—is explained next.

Eliciting Potential Enrollment in the IPP

The IPP section of the survey first described the program and its eligible areas in detail, accompanied by an easy-to-follow illustration.(The respondents who had been involved in pilot testing during survey design found a practical example of the program particularly illustrative and useful.) The eligible areas were described using wording and habitat classification that are common in forest management plans, forest management guidelines, and magazines and newspapers directed to forest owners.

Next, landowners were asked whether their forests included areas eligible for the program. Then, one at a time, three modified versions of the current IPP (Programs A, B, and C) were presented, and respondents were asked how many hectares of old-growth forests and biodiversity hotspots they would enroll in each program, if any. Each of the three participation questions addressed a different modification of the current IPP, defined by two attributes: payment per hectare enrolled (P, paid in a lump sum on enrollment) and contract length (T, in years, which specifies how long the enrolled land must be protected). Program attributes P and T were randomly varied across the respondents using the generic principles of statistically efficient experimental design.[v] The program participation question is illustrated in the Appendix 1.

The discrete continuous question format elicited enrollments in a way that landowners found both credible and easy to answer. Many of the respondents are full-time farmers who are familiar with the question format from land-use forms they complete annually to enroll farmland in the European Union's agricultural policy programs. Others are landowners who likely know their forests in detail. For example, about 90% of the respondents indicated that they personally are in practical charge of forest management, alone or with some assistance from the local forestry association (a public entity that provides guidance to forest owners). Among the small remainder of respondents, primary practical responsibilities were attributed to the local forestry association or some other party, typically a family member not listed as a property owner. The enrollment questions are of the same complexity as other recurring forest-management decisions, such as deciding whether to cut off, thin, plant, or otherwise manage an area of forest.

Econometric Modeling of Enrollment Responses

The goal of the econometric modeling was to predict program enrollment as a function of the key attributes of the program, landholdings, and landowners. Using estimation results and the characteristics of the candidate sites and their owners, we estimated the opportunity cost of conservation for each candidate site. In addition to modeling discrete participation choices, which is the approach chosen by most studies about enrollment in agricultural (Cooper 2003; Cooper and Keim 1996; Cooper and Osborne 1998; Lynch and Lovell 2003) and forestry (Langpap 2004) programs, we accounted for the amount of land enrolled by each participating landowner. Using the beta-binomial model, we estimated the probability of landowner *i* enrolling x_i hectares in program *t* as

$$p(x_{it} \mid n) = \binom{n_i}{x_{it}} \frac{B(v + x_{it}, n_i + w - x_{it})}{B(v, w)}, \quad Range\ 0 \le x_{it} \le n_i,\ x_{it} \text{ is integer}, v, w > 0 \tag{1}$$

where $B(v,w)$ is the beta function (where v and w are the estimated parameters), n_i is the number of hectares for each landowner, and p is a beta(v,w) random variable which will be modeled as functions of independent variables. The independent variables, their definitions, summary statistics, and scaling for estimation are presented in Table 1. The variable P is the per-hectare payment a landowner would receive for enrolling a hectare of biodiversity hotspots in conservation for T years.

Because Finland extends more than 1,000 kilometers from south to north, with considerable differences in the climatic and soil conditions across the country, we used information about growing season length, cumulative thermal sum, and other distinct regional characteristics to divide the country into four policy-relevant areas (South, Lakes, Central, and North [Figure 1]), which are included in the estimated models as indicator variables (SOUTH, LAKES, CENTRAL, and NORTH). The estimated model also includes several variables related to the landholding and the landowner:

- PRICE_EXPECTATION represents expected future timber prices relative to other prices on an ordinary scale of 1–5 (i.e., decrease strongly, decrease slightly, stay constant, increase slightly, and increase strongly); it is included because price expectations are heterogeneous and influence the cost of protection expected by landowners.
- TOTAL_LAND represents the size of the landholding, in hectares. The beta-binomial model will reasonably predict higher enrollments with greater land area even without TOTAL_LAND because it predicts total enrollment as proportional to the total size of the holding (see Appendix 2). TOTAL_LAND allows for the potential of an additional propensity toward enrolling land, which reflects a long-standing tradition in forest economics to examine how ownership structure and the fragmentation of forest holdings into smaller units affects forest management decisions.
- VOLUNTARY is an indicator variable equal to 1 if the landowner has already avoided harvesting a certain forest because of its valuable nontimber characteristics (e.g., natural attributes, landscape, or importance in family history); it is a proxy for the owner's in situ valuation of forests that can affect the compensation requirement for protection.
- NF_INCOME is the monthly nonforest income of the household (before taxes), shown both by conceptual (Tahvonen and Salo 1999) and empirical (Kuuluvainen et al. 1996) studies to potentially influence forest management decisions.
- SEX, which stands for the gender of the forest owner, and AGE, which is the age of the forest owner, are included based on the findings from empirical research related to the forest owners' management decisions (e.g., Kuuluvainen et al. 1996) and preferences for conservation programs (e.g., Langpap 2004).

The model was estimated using GAUSS and the maximum likelihood method. Before the estimation, we excluded from analysis 328 surveys whose respondents had indicated that their forests did not contain biodiversity hotspots. The resulting data included 801 surveys, each with enrollment responses to questions about three programs. We estimated $w_i = \exp(\beta_w y_i)$ and $v_i = \exp(\alpha + \beta_z z_i)$ to ensure that $v, w > 0$. The vector z_i consists of payment and contract length; y_i comprises other estimated independent variables of landowners i. Note that a $Beta(v,w)$ random variable is identical to a $Beta(w,v)$ random variable, so that the choice of whether v or w depends on covariates is not important. For the three programs, three probabilities are computed with the same underlying parameters, but the y and z data vary by program and by landowner, respectively.[vi]

The model estimates were mostly statistically significant coefficients with the expected signs (Table 2). Given that the model was estimated as $v = \exp(\alpha + \beta_z z)$, where z consists of payment and contract length, increases in v increase the likelihood of enrollment. Therefore, as expected, payment increases are positively associated with enrollment, and the required length of conservation decreases landowners' propensity toward participation.

The estimation results of coefficients estimated within w are interpreted as follows. Given the model structure, smaller values of the parameter $w = \exp(\beta_w y)$ are associated with increased propensity toward enrollment and thus lower the compensation required for a given enrollment. So, for example, the estimated regional dummies—which are all statistically significant—suggest that forest owners in the Lakes region are least likely to enroll, followed by forest owners from the Central, South, and North regions in increasing order of the likelihood of enrollment. This ordering is generally consistent with different land values across the four regions.

The estimated negative and statistically significant coefficient for SEX indicates that, ceteris paribus, female forest owners are more likely than their male colleagues to enroll in the proposed IPP. The positive and statistically significant coefficient for AGE suggests that old forest owners are less likely than young forest owners to enroll. The VOLUNTARY parameter has a statistically significant and negative coefficient, which suggests that landowners who have already voluntarily altered some of their forest management decisions because of objectives unrelated to timber production (e.g., landscape preservation or personal, historical, and natural significance of specific forest areas) are more likely to enroll than others. NF_INCOME is statistically significantly associated with enrollment, with its coefficient suggesting that the higher the nonforest income of the forest owner's household, the higher the likelihood of enrollment in the proposed program.

This latter finding was expected because, for example, the forest rotation model by Tahvonen and Salo (1999) suggests that when harvest and decisions about consumption and savings are linked, increases in nonforest income lengthen the optimal rotation period. Total land area of the forest holding and the future timber price expectations of the forest owners are not statistically significantly associated with potential enrollment in the proposed program. The statistical insignificance of price expectations is somewhat surprising but may be mostly due to the difficulty of measuring price expectations. The insignificance of the TOTAL_LAND variable indicates that, ceteris paribus, the amount of land enrolled is proportional to the amount of land available.

Using the estimated parameters, we predicted the minimum payment requirement for protection (i.e., opportunity cost) separately for each of the biodiversity hotspots for which we had biological data (see Appendix 2 for details). Our data allowed the estimation of the opportunity cost of conservation for 10–50 years without extrapolation beyond the experimental design. We chose 30 years of protection for our analysis because this period is commonly used by the agencies that prepare and assess forest conservation alternatives.

The estimated opportunity cost of protection is unique for each candidate site. The distribution of opportunity cost per hectare is skewed, with median, mean, and standard deviation estimated as $738, $6,861, and $14,348, respectively. The skewness of the distribution is logical, because even though many candidate sites are relatively inexpensive to enroll, some are highly valued for their standing timber volume or other characteristics (e.g., development potential for second homes). For example, the stumpage revenue at sites with substantial mature pine, spruce, or birch stands may exceed $10,000–15,000 per hectare.

In the following section, we exclude the most expensive 10% of sites from our analysis to keep the analysis within a range of scenarios that are realistically policy-relevant. The 10% cost threshold is ad hoc but mirrors that of actual conservation programs, which likely screen out the most expensive sites a priori.

Estimating Biological Benefits from Conservation

We estimated the biological benefits from the number of endangered vertebrate and invertebrate species supported by different types of biodiversity hotspots. This information was derived from two sources: biological assessments of the surveyed landowners' forests and field data on the primary habitats of endangered species.

Information about biodiversity hotspots within the forest holding is important to forest management decisions and often available in standardized assessments and forest management plans prepared by professional foresters. Although not every surveyed landowner listed this information, we obtained a subsample of 150 landholdings for which comprehensive data on existing biodiversity hotspots are available. The results of statistical tests indicated that the geographical distributions of these landholdings and the original sample were similar.[vii] A total of 507 hectares of biodiversity hotspots were identified; these areas constituted our candidate sites for protection.

Next, we determined biological benefit scores for each candidate site from the number of endangered vertebrate and invertebrate species supported by different types of biodiversity hotspots (listed in Table 3). The descriptions of different hotspot categories in the survey corresponded to those listed in Table 3 but went into more detail to facilitate the classification of hotspots by using common forest types. For example, habitats adjacent to watercourses were described as forests adjacent to streams, creeks, natural springs, rivers, and lakes, using the usual language in forestry. The data listed in Table 3 on the number of species supported by different types of biodiversity hotpots are originally from the field studies of SYKE, the Finnish Environment Institute (Rassi et al. 2001). These data can be matched with the biological assessments obtained from landowners in our survey, because our classification of biodiversity hotspots corresponds to that adopted by SYKE.

Ecological benefit scores were determined for each candidate site as follows. First, a biological benefit score was assigned to each habitat type l so that $h_l = \sum_{j=1}^{J} s_j$,

where s_j = 0 or 1 (1 indicates that habitat i provides primary habitat for endangered species s_j, $j = 1, \ldots, J$). Second, a biological benefit score $b_i = h_{li}$ was assigned for each candidate site i based on its habitat type l. The score was averaged whenever the site was classified as two or more habitat types. After standardization, the resulting benefit scores were 0.07–1.00 per hectare (median, mean, and standard deviation = 0.13, 0.20, and 0.205, respectively).[viii]

Clearly, no unique approach exists that can determine the nonmonetized benefits derived from protecting different habitat types. Ideally, one would predict the specific contribution to the long-term viability of the target species that resulted from the protection of each candidate site. Each site's contribution would depend on the rest of the program configuration, so a vast array of possible spatial configurations would have to be examined for each conservation target. Incorporating such dynamic and interdependent aspects of reserve decisions is one of the key current challenges in reserve design modeling (Williams et al. 2004) but is not yet feasible in most empirical applications.

Two primary causes contribute to this situation: (a) the lack of time-series population data necessary for modeling population viability and (b) the computational difficulties of incorporating even simplistic spatial dependencies in conservation reserve design problems, especially to those with many candidate sites. For example, we are aware of only two empirical studies that incorporate dynamic stochastic models of population viability and of reserve site selection: Westphal et al. (2003) combine a reserve site selection model based on stochastic dynamic programming methods with a population viability model of the emu wren, an endangered southern Australian bird, and Newbold et al. (2005) develop a stochastic model of salmon population dynamics and incorporate it into a reserve site selection framework for prioritizing the protection efforts for three salmon stocks in the Upper Columbia River basin. Although both of these studies deal with several orders of magnitude fewer species and an order of magnitude fewer candidate sites than in this application, they involve optimization problems under spatial dependencies that become so complex that only heuristic algorithms are available for identifying program configurations. So for the purposes of this study, and given that the policy problem is to safeguard the remaining habitat of endangered species, we consider the number of endangered species supported by different habitat types a logical and useful proxy for nonmonetized program benefits.

IV. Site Prioritization Using Different Program Variants

After estimating biological benefits and opportunity costs of protection at the site level, we examined alternative approaches to prioritizing hotspot conservation. First, consider n available candidate sites with a total area $\overline{X} = \sum_{i=1}^{n} x_i$. The protection of site i ($i = 1, \ldots, n$) provides a biological benefit b_i at the opportunity cost c_i (per unit area). Next, denote the total cost of protecting all n candidate sites $\overline{C} = \sum_{i=1}^{n} c_i$. For each conservation budget $M_j \leq \overline{C}$, the different program configurations are identified as follows.

- IPP: $Max \sum x_i$ so that $\sum c_i * x_i \leq M_j$.
- Species-only program: Select first site i for which $b_i \geq b_j$, $\forall j \neq i$. Continue while $\sum c_i * x_i \leq M_j$.
- Cost–benefit benchmark: $Max \sum b_i$ so that $\sum c_i * x_i \leq M_j$.

Spatial considerations, which are important when designing a conservation program for a large geographical area such as an entire country, can be examined by assessing programs with similar geographical coverage. Spatial constraints can be imposed across J regions (Figure 2) by obtaining the program configuration under the following constraint in each region j ($j, k = 1, \ldots, J$): $\frac{\sum x_{ij}}{\overline{\overline{X}}_j} = \frac{\sum x_{ik}}{\overline{\overline{X}}_k} \quad \forall k \neq j$.

V. Results

We used the range of observations in the data to normalize the cumulative costs and conservation benefits achieved by equal budgets with different programs (illustrated in Figure 2). The cumulative benefit score represents total conservation benefits.

By definition, the cost–benefit benchmark generates the greatest total biological benefits for any given budget (e.g., ~90% of maximum total benefits for ~20% of maximum total cost). The IPP is nearly as cost-efficient as the cost–benefit benchmark. Benefits of these two programs are almost identical up to ~85% of the maximum total benefits; thereafter, the IPP is slightly more costly. The species-only program clearly accumulates less benefit than the other alternatives until ~85% of maximum total benefits. Thereafter, all three programs are comparable.

We examined spatial considerations by dividing the country into three regions of approximately equal area—South/Lakes, Central, and North—and requiring each program alternative to cover an equal percentage of candidate areas in every region (Figure 3). (The same regions were applied in the estimation of enrollment functions; the South and Lakes regions are now joined so that each region had an approximately equal number of biodiversity hotspots for comparison.) Spatial constraints moderately alter the relative advantage of IPP and cost–benefit programs, but the IPP continues to generate considerably higher benefits than the species-only program up to ~80% of maximum total benefits.

The relative cost of achieving different conservation targets varies between program alternatives and target conservation levels (Table 4). For example, when excluding spatial constraints, the cost of generating 60% of the maximum benefits by using the IPP is 8% of the cost of reaching the same target by using the species-only program. Among spatially balanced programs, the IPP generates 60% of total benefits for 23% of the cost of the species-only program. Overall, the IPP achieves most conservation targets more cost-effectively than the species-only program does.

Figure 4 illustrates the potential country-level cost savings of using the IPP relative to the species-only program, assuming that the sample of hotspots is representative. Although this assumption is valid based on the cross-regional distribution of our sample, assessing these hotspots relative to their other characteristics is not feasible. For this reason, we present country-level results primarily to illustrate the scale of potential cost-efficiency gains. For conservation targets of 30–80% (a likely range for actual policies), the IPP can save ~$200–700 million. Potential savings of using the IPP do not appear to be very sensitive to assumptions about the programs' spatial configuration.

Finally, we examined the sensitivity of the results relative to the estimation of the biological benefits and the opportunity costs of conservation. First, only relative benefits and costs affect site prioritization and the program configuration. For example, because absolute benefits and costs are not pertinent, one can divide or multiply all costs by the same constant without distorting the site prioritization in the cost–benefit benchmark or the IPP. Therefore, systematic under- or overestimation of the benefits or of the opportunity cost of conservation would not alter the prioritization of candidate sites.

Second, we examined the effects of altering model specification in the estimation of enrollment function on the relative cost-efficiency of the IPP and the species-only program. Results of these examinations clearly suggest that the prioritization of candidate sites—thus the relative cost-efficiency of IPP and species-only programs—is robust to alternative model specifications in the estimation of the opportunity cost of conservation.

Third, our measure of biological benefits focused on invertebrate and vertebrate species and weighed each species equally, even though other weighting schemes could have been considered. Therefore, we examined the sensitivity of the results to including other taxonomic groups in the biological benefits function and found that doing so did not alter our main result: the relative cost-efficiency of alternative programs. This finding is logical because the richness of endangered species across different taxonomic groups at different habitat types is positively correlated, causing different taxonomic groups to lead to similar prioritizations of candidate sites.

VI. Discussion

This study assesses and highlights the potential of using IPPs for protecting biodiversity in private nonindustrial forests. A fairly simple IPP—whereby landowners enroll eligible tracts of land—can achieve conservation targets with surprising cost-efficiency. The IPP performs better than species-only site selection and is nearly as efficient as the cost–benefit benchmark, the most cost-effective but hypothetical solution to the conservation program. The gains from using an IPP are especially dramatic when only a fraction of all candidate areas are protected. Given the economic and political realities of endangered species protection, this scenario is likely reasonable in practical policymaking.

Conservation advocates and professionals are sometimes skeptical toward the use of IPPs or, more generally, market-based approaches to biological conservation. Some of this skepticism may relate to the resistance to allowing “market forces” to determine part of conservation program configuration. To some degree, this concern is understandable. In pure top-down programs, for example, the regulator may be able in principle to configure a program that is spatially more optimal than would result from an IPP. But even if the top-down approaches have some advantages over an IPP (e.g., by enabling conservation biologists to freely choose their preferred mixture of protected habitat), one should determine whether those advantages justify the extra costs. In addition, as the spatially explicit program configurations demonstrate in this paper, IPPs can also facilitate refined program arrangements, possibly with cost-efficiency advantages similar to those discovered in this analysis.

Different information requirements across programs are also extremely important in practical policy applications. The cost–benefit benchmark requires the most information: data on biological benefits and the opportunity costs of conservation, which (especially the latter) are typically unobservable to conservation agencies. The species-only program requires information about the biological characteristics of candidate sites. At least in principle, this information is possible to collect in Finland, where (unlike in the United States) the right of public access to both public and private land (*jokamiehenoikeus*) is a key convention of property rights (similar to *allemansrätten* in Sweden and *allemannsretten* in Norway). But in practice, information gathering costs money and time, both of which are central constraints in the protection of endangered species. IPPs require no site-level information but rely on a marketlike sorting mechanism whereby the opportunity cost of conservation determines enrollment. Some data are needed to design and evaluate an IPP, but these requirements are modest overall relative to those of other programs. Thus, concerning the information requirements, IPPs seem at least as good as—if not better than—top-down programs.

Each conservation problem is different; no set of results will guarantee what will be obtained in other situations. Because the heterogeneity of and correlation between conservation benefits and costs determine the relative efficiency of alternative programs, the costs and benefits of alternative conservation programs should be scrutinized, case by case, before determining which alternative best meets the overall objectives of a conservation policy under evaluation. Also, like any other conservation programs, voluntary programs must be properly enforced to be successful. However, we expect that IPPs often generate more cost savings and accumulate more conservation benefits than other commonly used methods (e.g., species-only criteria) by tacitly capitalizing on the private knowledge of landowners about the opportunity costs of conservation.

Although this study highlights the need to consider landowner behavior in the assessment of bottom-up conservation programs, our results do not suggest abandoning the use of biological criteria in designing and implementing conservation programs. Because every candidate site examined in our analysis was prescreened using biological criteria, the results highlight the importance of combining IPPs with biological criteria.

Biological criteria help determine which sites are of interest to conservation; the IPP's marketlike policy mechanism helps prioritize them cost-effectively.

Appendix 1. Participation Question Example

Program A	
Payment	*P* per hectare
Contract Length	*T* years

How many hectares would you enroll in under the conditions of Program A?

I would enroll the following areas:

(a) ☐ ☐ ☐ hectares of old growth

(b) ☐ ☐ ☐ hectares of biodiversity hotspots

(c) ☐ I would not enroll anything.

Appendix 2. Predicting the Opportunity Cost of Conservation

Consider the total amount of land enrolled when a contract with common payment *P* and contract length *T* is offered to all landowners. According to the beta-binomial model, the expected number of hectares enrolled by landowner *i* is

$$\bar{X}_i = N_i\left(\frac{v_i}{v_i + w_i}\right) \qquad \text{(A2-1)}$$

where N_i is the total hectares in holding *i* and v_i and w_i vary by landowner. For each landowner, different combinations of *P* and *T* yield different expected enrollments. When $v_i = \exp(\alpha + \beta_T T_i + \beta_P P_i)$, solving Equation A2-1 for P_i gives the marginal cost of protecting $\bar{X}_i$ hectares:

$$P_i = \left[\left(\frac{w\bar{X}_i}{N_i - \bar{X}_i}\right)\left(T_i^{-\beta_T}\right)\left(\exp^{-\alpha}\right)\right]^{\frac{1}{\beta_P}}. \qquad \text{(A2-2)}$$

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Tables and Figures

Table 1. Variable Descriptions and Summary Statistics

Variable	*Description*	*Mean (FIM)*	*Standard Deviation*	*Scaling in Estimation*
P	Payment (lump sum)	21,941.6	15,579.1	ln(*P*/1000)
T	Contract length (years)	28.3	13.1	ln(*T*/10)
SOUTH	Dummy: South region	0.23	0.42	-
CENTRAL	Dummy: Central region	0.19	0.39	-
NORTH	Dummy: North region	0.38	0.49	-
LAKES	Dummy: Lakes region	0.20	0.40	-
PRICE_EXPECT	Forest owner's expectation of future timber prices (*1–6*)	3.69	1.48	-
TOTAL_LAND	Size of the forest holding (hectares)	44.88	68.09	ln(TOTAL_LAND/10)
VOLUNTARY	Dummy: landowner has avoided logging or other management due to nontimber objectives	0.21	0.41	-
SEX	Gender of the forest owner (0 = man, 1 = female)	0.76	0.43	-
NF_INCOME	Monthly nonforest income of the forest owner (FIM)	10,028.7	7,775.6	ln(NF_INCOME/3000)
AGE	Age of the forest owner (years)	59.2	13.28	ln(AGE)

Note: FIM = Finnish markka (1 FIM = $0.15).

Table 2. Beta-Binomial Model Results

Coefficient	*Estimate*	|*t-statistic*|
α	–4.6014***	17.52
P	0.4210***	4.391
T	–0.4873***	3.080
SOUTH	–5.6052***	2.905
CENTRAL	–5.2742***	2.749
NORTH	–6.3541***	3.297
LAKES	–4.3380**	2.187
PRICE_EXPECTATION	–0.1166	0.823
TOTAL_LAND	0.0957	0.846
VOLUNTARY	–0.6322**	2.402
SEX	–0.8270***	3.270
NF_INCOME	–0.6100***	3.601
AGE	1.5566***	3.374
Max log-likelihood	–1,028.75	
Observations	801	

** and *** indicate parameter significance at 0.05 and 0.01 levels, respectively.

Table 3. Primary Habitat of Endangered Species, by Taxonomic Group

Primary Habitat	*Vertebrates*	*Invertebrates*	*Vascular Plants*	*Cryptogams*	*Fungi*	*Total*
Old heath land, herb-rich forests	5	127		9	102	243
Other heath land forests	3	10	3	1	24	41
Herb-rich forests (meadows)	2	64	26	4	120	216
Esker forests		15	6			21
Forest fire areas		29				29
Spruce mires, fens		5	17	22	6	50
Other wetlands	1	9	1	3	3	17
Human-made environments		31	10		22	63
Watercourses, shoreline areas, etc.	25	146	48	30	16	265
Other forests	2	7		1	4	14
Total	38	443	111	70	297	959

Note: Vascular plants, cryptogams, and fungi are not used in the biological benefit estimations; they are presented here for comparison.

Table 4. Cost of Achieving Conservation Targets, Relative to Species-Only Program

Benefit Target	*Spatially Unconstrained Program*		*Spatially Balanced Program*	
	IPP	*Cost–Benefit*	*IPP*	*Cost–Benefit*
10%	0%	0%	2%	2%
20%	0%	0%	4%	4%
30%	2%	1%	5%	5%
40%	4%	3%	8%	6%
50%	7%	3%	16%	13%
60%	8%	6%	23%	24%
70%	18%	15%	28%	26%
80%	34%	30%	69%	43%
90%	74%	40%	110%	62%
100%	100%	100%	100%	100%

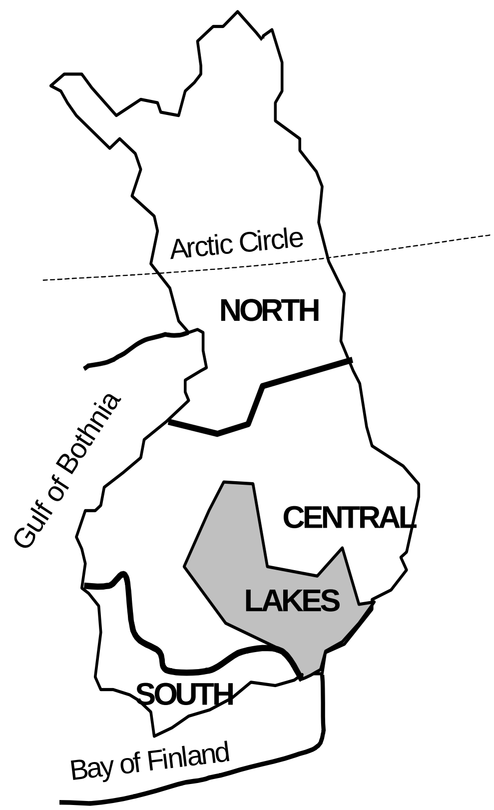

Figure 1. Model Regions

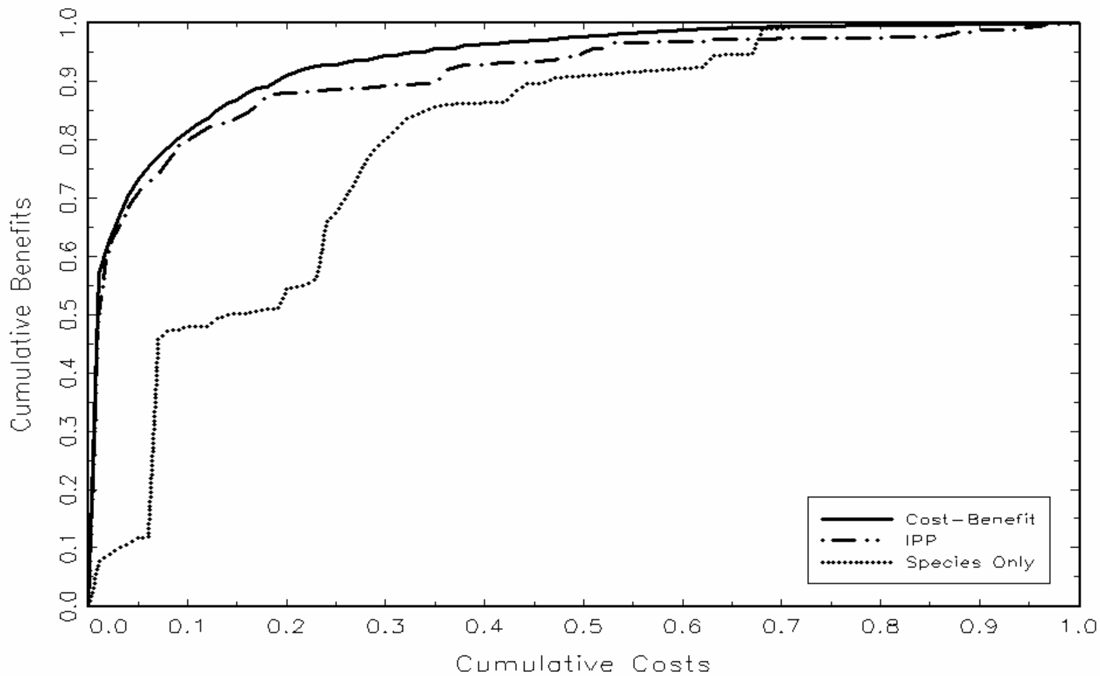

Figure 2. Costs and Benefits of Alternative Programs

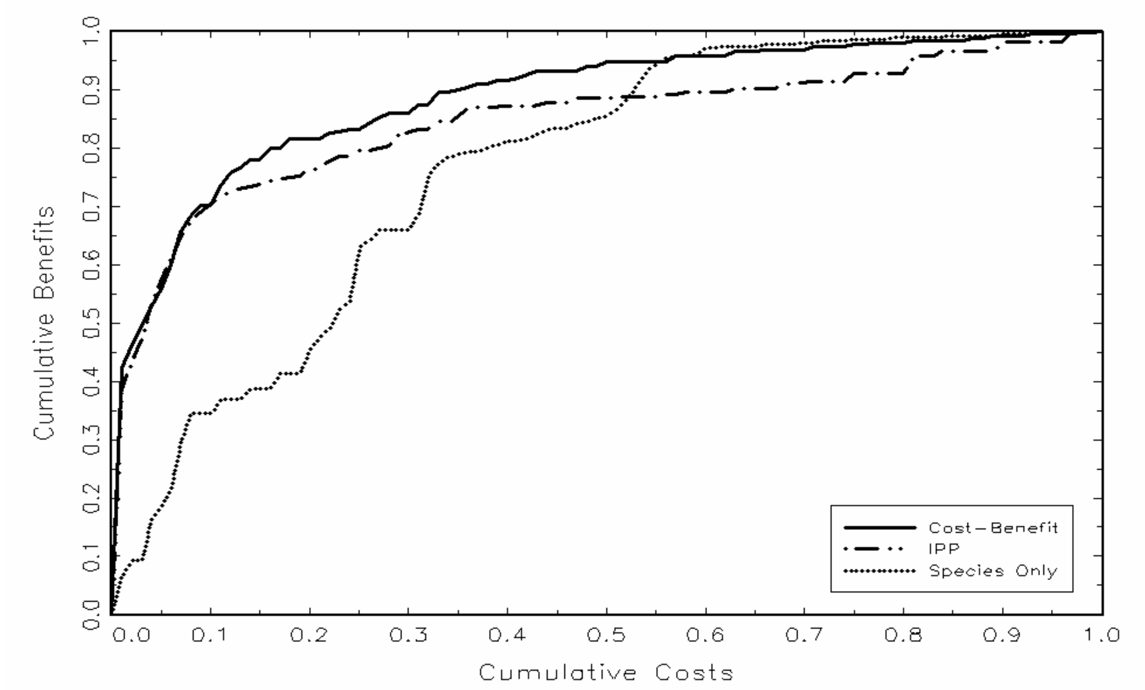

Figure 3. Costs and Benefits of Alternative Spatially Explicit Program Configurations

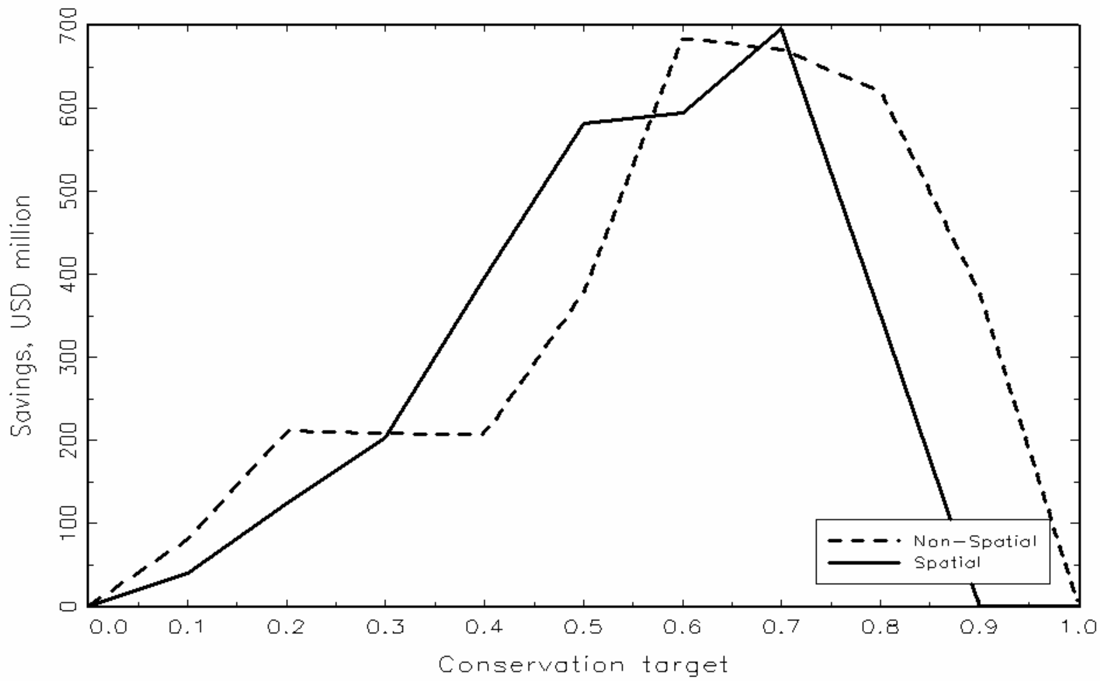

Figure 4. Potential Savings (in US$) of the IPP Relative to Species-Only Program

Notes

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[i] Agricultural IPPs are sometimes assessed using biological assessments of enrolled areas (e.g., van Buskirk and Willi 2004). However, these assessments are snapshots of protected land that do not examine the effectiveness of IPPs relative to other possible approaches to protecting biodiversity, such as traditional top-down regulatory programs.

[ii] The IPP represents a habitat-based approach to conservation (e.g., Hughes et al. 2000). Understanding the performance of habitat-based approaches relative to species- or population-based conservation approaches is a central question in conservation biology but outside the main scope of the current study.

[iii] We used a stratified sampling scheme with province-specific quotas (250–350 forest owners in each of the 10 provinces) to guarantee representative cross-regional data.

[iv] Of all surveys mailed, 20 were returned due to discontinued ownership of forests by the respondent (e.g., due to property sale or death).

[v] Each respondent was always offered a higher payment for longer contract lengths and not vice versa. Although necessary in this case, using a correlated design can lower efficiency in econometric estimation (e.g., Huber and Zwerina 1996). To minimize this possibility, we created an array of randomly drawn alternative survey designs, each of which satisfied the above nondominance constraints. The program ranked alternative designs according to the level of nonorthogonality. The final design was chosen from among the 200,000 least nonorthogonal designs that satisfied the nondominance constraints. The incentive payment in different programs was 500–70,000 Finnish markka(~US$100–$14,000) per hectare. The contract length was 10–50 years, in 5-year increments.

[vi] Previous empirical research on landowners' willingness to participate in environmental policy programs has focused on the use of discrete choice econometric models to explain landowners' willingness to participate in various environmental and agricultural policy programs (e.g., Cooper and Keim 1996, Cooper and Osborne 1998, Cooper 2003, Lynch and Lovell 2003, and Langpap 2004); each of these studies models program participation as a binary choice. In another application, Wu and Babcock (1998) used a multinomial logit model to examine farmers' choice of tillage, rotation, and soil-testing practices. We have also considered flexible multivariate censored regression models, which nest the discrete choice econometric models estimated in the studies listed above. Our results (not reported here due to space limitations) indicate that although the beta-binomial model has fewer parameters to estimate, it performs either as well as or better than the censored regression model. One last approach worth noting is the use of switching regression models for addressing sample selectivity problems that can arise when examining

actual participation and management choices by farmers (Fuglie and Bosch 1995). Sample selectivity issues are not important in our application because we collected survey data from a random sample of forest owners.

[vii] The forest owners were distributed across the four regions as follows (150 respondents in the hotspots sample in parentheses): South 21% (20%), Central 33% (34%), North 30% (29%), and Lakes 17% (17%).

[viii] The following example explains the calculation of benefits scores. The total number of endangered invertebrate and vertebrate species is 481 (443 + 38; Table 3). Habitats adjacent to watercourses provide the primary habitat for 171 species, therefore, h = 171 for such habitats. Biological benefit scores are calculated similarly for other habitat types, then standardized relative to the highest score (171). So, for example, the standardized score for meadows is approximately 0.39 (calculated as 66/171).