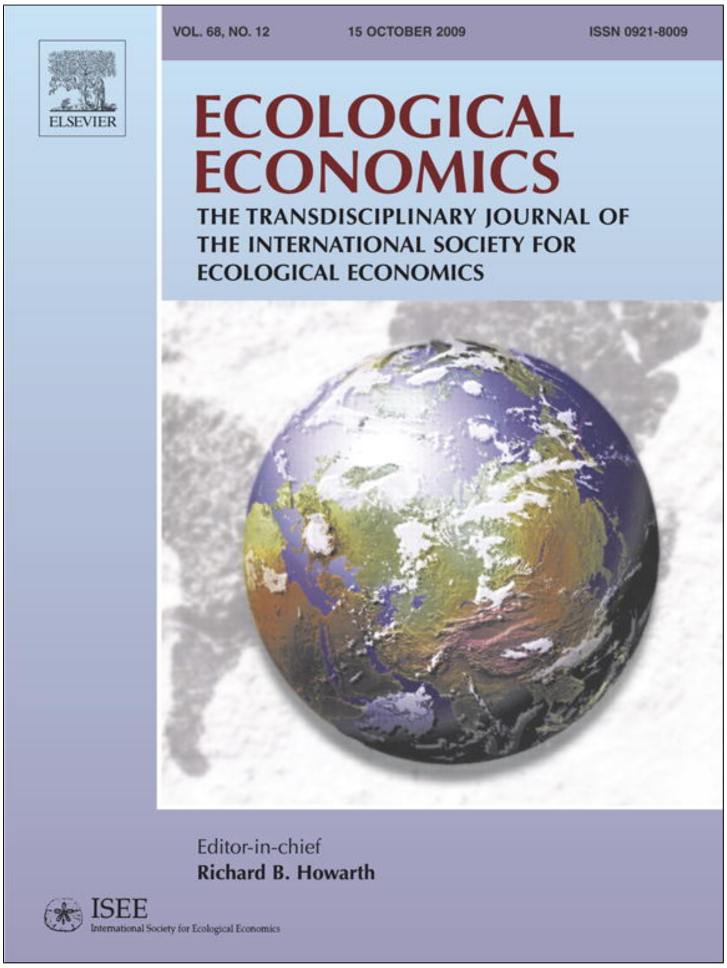
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Analysis

Does a voluntary conservation program result in a representative protected area network? The case of Finnish privately owned forests

Artti Juutinen [a,*], Mikko Mönkkönen [b], Anna-Liisa Ylisirniö [c]

[a] Department of Economics, P.O. Box 4600, FIN-90014 University of Oulu, Finland
[b] Department of Biological and Environmental Science, P.O. Box 35, FIN-40014 University of Jyväskylä, Finland
[c] Arctic Centre, University of Lapland, P.O. Box 122, FIN-96101 Rovaniemi, Finland

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ABSTRACT

Conservation contracting has attained growing interest worldwide as a tool for protecting biodiversity in privately owned lands. In this policy, landowners receive payments from an environmental agency in exchange for land use practices that contribute to the supply of biodiversity. This approach may result in a conservation network which does not cover all focal ecological characteristics, because landowners determine the supply of potential targets. In addition, the contracts are typically allocated by using a scoring method that is not giving information on the representativeness of the species composition of the sites. In this study, we investigated what is the performance of a voluntary conservation program in selecting sites that would maximize the number of specific target species in the selected conservation network subject to a given budget constraint. We focused on the Finnish pilot program named Trading in Natural Values (TNV). Our data consisted of 56 mature stands covering both stands that were offered to the TNV program and stands that were not offered. All the stands were surveyed for specific groups of wood-inhabiting fungi and epiphytic lichens that can be considered as good surrogates for forest species diversity.

Our results showed that the participation in the TNV program was large enough to meet the ecological goals, because the offered targets uncovered only two of the 73 surveyed species, and the cost-effective conservation network included only a few targets that were not offered in the pilot program. However, the contract allocation method used in the TNV could be improved, because many ecologically valuable targets that were offered to the program were not accepted. In general, it could be justified to survey some indicator species, which would be relatively easy to identify, to maximize species coverage in contract allocation. Surveying indicator species causes some extra costs, but these are likely to be minor compared with the costs savings in opportunity costs, due to the improved targeting of protected areas.

1. Introduction

Concerns on the effects of land use on the supply of environmental amenities in private lands have led worldwide to increasing demand for landowners' voluntary participation in conservation programs. This type of policy is based on conservation contracting, where landowners receive payments from an environmental agency (e.g. government) in exchange for land use practices that contribute to the supply of an environmental amenity, such as clean water or biodiversity.

In particular, conservation contracting has been used in many agri-environmental schemes in which landowners are paid to adapt their operations to benefit biodiversity among other things. Many endangered habitats and species require certain land use practices in order to survive. Another conservation policy has been to establish reserves through government land acquisition. Ultimately, the former can be seen as a market mechanism and the latter as a top–down regulation. Examples of agri-environmental schemes used in Europe include English Environmental Stewardship Scheme (http://www.naturalengland.org.uk/ourwork/farming/funding/es/default.aspx) and the German MEKA program (Wilson, 1995). The Habitats Directive and the Birds Directive (Council Directive 79/409/EEC) are the two most important instruments for establishing reserves in Europe. They are designed to create a coherent network of protected areas known as NATURA 2000.[1]

* Corresponding author. Fax: +358 8 5532906.
E-mail address: artti.juutinen@oulu.fi (A. Juutinen).

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[1] European conservation policy has applied fixed payments (or individually negotiated grants) this far. Only in few pilot programs, conservation contracts have been allocated using a competitive bidding mechanism, such as auctions where several landowners offer simultaneously their land into the program by delivering bids to the authority of program who accepts the bids as "value for money" basis until the budget available for the program is exhausted. Conservation auctions have been tested, for example, in the state of North Rhine-Westphalia, Germany, for the maintenance of low-intensity grazing systems (Holm-Müller and Hilden, 2004).

To conserve biodiversity on private forestlands, Finland tested in 2003–2007 a voluntary conservation project called Trading in Natural Values (TNV) in south-western Finland (Gustafsson, 2008). The aim of TNV was to create markets for biodiversity in a manner that has a broad acceptance in society and particularly among forest owners.[2] Drawing on the experience of this program, the Finnish government has recently decided to apply this intensive mechanism to all of Southern Finland, where about 2% of forestland is currently protected (Government Resolution, 2008).

Using voluntary conservation contracting for biodiversity has many favourable features (Segerson and Miceli, 1998). Conservation contracting, based on competitive bidding, has the potential to reveal landowners' opportunity costs, thereby reducing the information asymmetry between the landowner and environmental agency. Another important feature is that conservation contracting acts as a price discovery mechanism for non-market environmental goods and services. Thus, using these sorts of quasi-markets may improve the efficiency of biodiversity conservation compared with the centralized conservation policy where an environmental agency selects the sites to be included in reserves and pays fixed payments to landowners for protecting their lands.

One possible problem in conservation contracting for biodiversity is that the resulting conservation network may not cover all focal ecological characteristics. This can happen because landowners determine the supply of potential targets. Regarding biodiversity, ecologically valuable sites may have features that can rarely be found on other sites, indicating that some sites are irreplaceable and there are no close substitutes for them. Thus, the supply of certain biodiversity benefits may be short or even non-existing depending on whether the owners of these sites are willing to participate in the conservation program or not. This feature suggests that a voluntary conservation program may not reach the given ecological targets or it can turn out to be costly from the social point of view.

It is important also to notice that some sort of indexing is typically used to determine the ecological quality of the offered targets in the conservation contracting programs. For example, the Conservation Reserve Program (CRP) in the U.S.[3] employs an environmental benefit index (EBI) to compare bids (see Latacz-Lohmann and Schilizzi 2005 for a detailed description of CRP and several other case studies of conservation auctions). In the TNV program the implementing agency of the programme assessed the ecological quality of the offered targets by using the guidelines determined by the Ministry of the Environment (Kriteerityöryhmä, 2003). The guidelines emphasize certain ecological characteristics, such as large-diameter broadleaved trees and pines, dead or burned trees, threatened species, luxurious vegetation, natural water conditions, distance to existing nature protection areas, size of the area, and landscape values. This sort of scoring procedure may not always be ecologically efficient (Pressey and Nicholls, 1989). For example, Frank and Müller (2003) concluded that in Austria a voluntary forest conservation program was not successful in selecting rare and vulnerable forest communities. Mönkkönen et al. (2009) showed, however, that the procedures used in TNV during site selection and negotiations were appropriate and non-opportunistic from an ecological viewpoint, being able to select sites which were valuable in their species composition.

In this study we assess the cost-efficiency of TNV. In particular, we analyse whether or not TNV resulted in a representative conservation network, i.e. the network encompassing the full spectrum of focal ecological values. The representativeness affects the extent to which voluntary conservation must be supplemented by more traditional conservation methods such as compulsory land acquisition. Thus, although we examine the performance of the Finnish pilot program, the issue in this paper has global importance.

More precisely, we investigated what is the performance of a voluntary conservation program in terms of selecting sites that would maximize the number of species in the selected conservation network subject to a given budget constraint. To answer this question we used empirical data on 56 mature stands. These stands represented four groups: 1) stands protected in TNV (TNV sites). 2) Stands which were offered by landowners to the TNV program, but which were not negotiated due to the lack of natural values judged during the preliminary field survey (typical managed forests, MF sites). 3) Stands which were offered to TNV and negotiated, but no agreement was made (compensation disagreement, CD). 4) Ecologically valuable stands which were not offered to TNV (potential sites, PS).

All 56 stands were surveyed for selected groups of wood-inhabiting fungi and epiphytic lichens. Wood-inhabiting fungi are dependent on dead wood and they are considered good indicators of dead-wood continuity and naturalness of a forest area (Bader et al., 1995). Some 6000–7000 species in Fennoscandia depend on dead-wood habitats (Stokland et al., 2004). Species that are dependent on dead wood represent alone 20% of all the threatened species in Finland (Rassi et al., 2001). Occurrence of epiphytic lichens has also been proposed as an indicator of forest continuity and conservation value in boreal forests (Kuusinen, 1995; Esseen et al., 1996). Several epiphytic lichen species are confined to old-growth habitats with long continuity (e.g., with old living trees), and their biomass tends to be considerably higher in old-growth than in managed forests (McCune, 1993; Esseen et al., 1996). Thus, the inventoried groups can be considered as good surrogates for forest species diversity.

Moreover, we estimated the opportunity costs of conserving these stands using information on their harvest and land values. The opportunity cost of landowners is, however, unobservable private information, i.e. harvest and land values (approximating forgone income from the land in its original use) do not reflect the actual preferences of landowners (Stavins, 1999; Levins and Plantinga, 2007; Nelson et al., 2008). In particular, environmentally minded landowners may be willing to protect their lands with a compensation that is lower than the market price based compensation (Michael, 2003). In order to take into account this aspect we alternatively estimated the opportunity costs using information on the observed rental payments paid in TNV and conducted a sensitivity analysis of our results.

We used a standard budget constrained site selection model in the analysis (Ando et al., 1998; Polasky et al., 2001; Juutinen et al., 2004). The solution of the budget constrained model served as a cost-effective benchmark to which we compared the practice used in TNV in order to reveal the performance of conservation contracting. The ultimate aim of the TNV program was to encourage forest owners to produce ecological values on their land and thereby provide a cost-efficient tool to halt species endangerment in commercial forest landscapes. Therefore, this comparison is highly relevant. In addition, following Malcolm and ReVelle (2005), we developed a site selection model based on species multiple representations in the system of reserves to take into account species persistence in prioritizing conservation targets.

As far as we know, it has not been examined earlier whether a voluntary conservation program results in a representative network in

[2] The traditional conservation policy in Finland has been the government land acquisition to establish conservation networks. In this mechanism, the government first identifies the ecologically most valuable targets, and then purchases the lands from private landowners. If the landowners have not been willing to sell their lands voluntarily, the government has used land takings. Consequently, many landowners may have a negative attitude toward nature conservation and do not support any further conservation efforts by the government (Matinaho et al., 2005).

[3] Regarding the United States, examples of the agri-environmental schemes include the Environmental Quality Incentives Program (EQIP) by the USDA and the Private Stewardship Program by the U.S. Fish and Wildlife Service. Auctions have been used in CRP and EQIP, for instance (Johansson, 2006).

the context of cost-efficiency.[4] Juutinen et al. (2008a) investigated the performance of TNV by comparing land purchase and leasing options. It was shown that land purchase and leasing yielded quite similar cost levels in the long run, which indicates that the competitive bidding process in TNV has not worked properly. To facilitate the consistent cost comparison they assumed, however, that the conserved sites were the same both in land leasing and purchasing. In this study, we relax this assumption to address the question of optimality of the selected network. Siikamäki and Layton (2007) examined the potential cost-effectiveness of incentive payment programs relative to traditional top–down regulatory programs for biological conservation in forestland. Their conclusion was that the incentive payment programs may be considerably more cost-effective than traditional top–down regulatory programs. Siikamäki's and Layton's study is based, however, on survey data, not on actual contracts.

2. Methods and material

2.1. Description of TNV

TNV was run in 2003–2007 in Satakunta in south-western Finland (Fig. 1). The pilot program was based on 10 year-long contracts between the private landowners and the government. According to these contracts, the forest owners produce biodiversity services in their lands and receive a compensation payment. The South-West Finland Forestry Centre acted as the representative of the government. It called for the bids from private landowners and negotiated on the rental payment and services provided by the landowner. Typically, harvesting is prohibited during the contract period, but in some cases the contract requires that the landowner improves the ecological quality of the stand, for instance, by artificially creating dead wood. The annual budget for TNV was about € 400,000, and 158 biodiversity conservation contracts were signed during the pilot program. The acceptance rate of bids was 44%. The main reason for the rejection of the bids was low ecological quality of supplied stands. However, in some instances there was disagreement on the proper rental payment (Gustafsson, 2008).

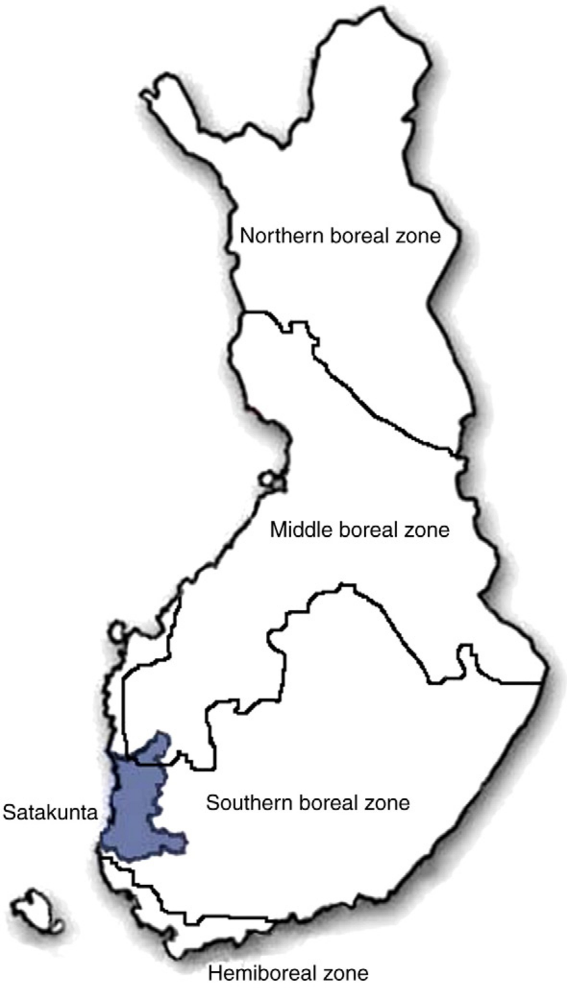

Fig. 1. Map of Finland with vegetation zones, study region (Satakunta) is shown in grey.

2.2. Site selection models

Consider a given geographical area with n different mature stands that foster m species. Each stand represents one site that possibly can be added to the conservation network. Denote the status of stands by x_j ($j=1\ldots n$), which gets a value 1 if the stand is selected into the conservation network and 0 otherwise. Let c_j be the opportunity cost of conserving the stand j, i.e. the opportunity costs vary between the stands. The overall conservation budget is C.[5] From a social point of view, the problem is to decide which stands to select into the conservation network, because all candidate stands can not be protected due to the limited conservation budget.

Suppose an environmental agency seeks to maximize species richness in the selected conservation network subject to a given budget constraint, i.e., the agency selects the stands cost-effectively. Denoting the index and set of species by h and the subset of candidate reserve stands that contains species h by N_h, the budget constrained site selection model (named as the basic model, hereafter) can be expressed as follows:

$$\max_{x_j, y_h} \sum_{h=1}^{m} y_h \tag{1}$$

s.t.

$$\sum_{j \in Nh} x_j \geq y_h \quad h = 1, \ldots, m \tag{2}$$

$$\sum_{j=1}^{n} c_j x_j \leq C \tag{3}$$

$$x_j, y_h = (0, 1) \ \forall i, h. \tag{4}$$

The target function (1) sums the number of species in the selected stands. Constraint set (2) ensures that species h is counted as being represented when at least one of the stands where it occurs is selected. The budget constraint is given in (3). The constraint set (4) simply indicates that the choice variables must be binary; the stands are either protected or rejected, and the species are represented in their entirety or not at all.

Importantly, the above model includes the concept of complementarity (May, 1990; Vane-Wright et al., 1991), which measures the contribution that an area, or a set of areas, makes to an existing network of reserves in terms of unrepresented natural features (Margules and Pressey, 2000). Thus, the basic model selects stands

[4] There exists an extensive body of theoretical literature concerning the efficiency of voluntary agreements in nature conservation (e.g. Stranlund, 1995; Segerson and Miceli, 1998; Wu and Babcock, 1999; Smith and Shogren, 2002; Langpap and Wu, 2004). Previous empirical studies related to this issue have examined, for example, the performance of different pricing rules in conservation reserve programs, but enrolled land areas were treated as having equal ecological quality in these analyses (Smith, 1995; Whitby and Saunders, 1996).

[5] Conservation causes also inventory costs (Juutinen and Mönkkönen, 2004) and transaction costs (Juutinen et al., 2008a). We discuss these issues in the result section.

that supplement each other from the perspective of species richness and in this sense takes into account the interdependence between stands. This ensures that the reserves are chosen according to their representativeness, that is, according to the extent with which the focal natural features occur in the reserves (Faith and Walker, 1996). If the stand interdependence in providing biodiversity is neglected in the site selection, the resulting conservation network may be biased in the sense that some natural features are present in several protected areas and some other focal features are totally missing (Pressey and Nicholls, 1989).

The opportunity cost of conservation for each stand can be determined by assessing the lost profits due removing the land permanently from timber production, i.e. using the Fautsmann model (Faustmann 1849). This procedure yields an approximation on market price based payment (i.e. asset value) paid to the landowners for conserving their land, and therefore, reflects the practice used in the traditional Finnish conservation policy, for example. It does not take into account landowners' preferences, however. In conservation contracting landowners' actual behaviour might not be well predicted by the asset value. Alternatively, the opportunity cost can be determined by using the observed rental payments paid in the voluntary program. Following Juutinen et al. (2008a), the Eq. (5) shows how the rental payments can be put into a comparable asset values.

$$c_j = \frac{P_j}{1 - e^{-rt}}, \tag{5}$$

where P_j denotes the lump-sum rental payment paid to the landowner in the beginning of the contract period to conserve stand j, r is the real interest rate, and t is the length of contract period. Thus, in right-hand side of Eq. (5) the denominator implicitly indicates that the contracts are infinitely renewed with t year intervals. In this study, Eq. (5) can be used only for the stands belonging into the TNV group, because the information on rental payments is not known for the other stands. For this reason, we use the Eq. (5) only in sensitivity analysis to consider robustness of our results that base on the asset values.

We use the basic model here to determine the optimal conservation network, and compare this outcome with the network chosen in TNV. For that purpose, we set the conservation budget C to the level equalling the sum of opportunity costs of protecting the same stands that were protected in TNV. It is interesting to detect the difference in the number of species between the cost-effective site selection strategy and the strategy applied in TNV. In addition, we vary the budget level to find out the trade-off between the number of protected species and conservation costs. We apply this procedure for the set that includes all the stands, and for the set that includes only the TNV stands to compare the performance of a cost-effective site selection and protection of the TNV stands. The latter represents the current conservation policy in a sense that it is based on the TNV stands. However, in the TNV pilot program, the stands were not selected to maximize the number of species as the information on species presence was not available. Instead, the stands were selected using information on structural characteristics of offered targets. This procedure is close to a scoring based on a benefit–cost ratio. We define the B/C ratio as:

$$\mathrm{B/C\,ratio}_j = \frac{\mathrm{EBI}}{c_j} \quad j = 1, \ldots, n, \tag{6}$$

where EBI denotes the environmental benefit index describing the ecological quality of the considered target (as will be described in the next section). In this procedure, the stands with the highest B/C ratio are selected first in the conservation network. In what follows, we compare also this scoring approach with the cost-effective conservation.

The basic model does not take into account species survival in the long run. This is a difficult issue to deal with in a site selection approach (Cabeza and Moilanen, 2001). It is clear, however, that a species found in several protected stands will survive more likely than a species found in a single stand. According to this simple principle, we next develop a site selection model (referred to as the backup model, hereafter), which takes into account species' multiple presentations. A species is said to have backup representation in the system of reserves if it is covered, or represented, in two or more stands (Malcolm and ReVelle, 2005).[6] In addition to the previous notation, let us denote species backup coverage by b. This backup can vary from 1 to k. If $k = 2$, for example, the backup model aims at protecting species found in a single stand (primary coverage) along with species found in two protected stands (backup coverage of two stands). More precisely, we assume that the benefits from protecting a species that has not yet been found in any stand in an existing conservation network are equal to the benefits from getting a backup coverage (of two stands) for a species that has been found in a single stand in an existing conservation network. The backup model is as follows:

$$\max_{x_j, y_h} \sum_{b=1}^{k} \sum_{h=1}^{m} y_{bh} \tag{7}$$

s.t.

$$\sum_{j \in Nh} x_j \geq \sum_{b=1}^{k} y_{bh} \quad h = 1, \ldots, m \tag{8}$$

$$y_{(b+1)h} \leq y_{bh} \quad b = 1, \ldots, k-1 \tag{9}$$

$$\sum_{j=1}^{n} c_j x_j \leq C \tag{10}$$

$$x_j, y_{bh} = (0, 1) \quad \forall i, bh. \tag{11}$$

The target function (7) sums the backup coverage of species in the selected stands. Constraint set (8) ensures that species h is counted as being represented as many times as it is found in selected stands. If a species is found in a single protected stand, it is counted once in the target function, and if a species is found in two protected stands, it is counted twice and so on. Constraint set (9) ensures that the counting of species representation is done in a logical order. A species is counted as being represented when it is found in single stand. After that, it is counted as having a backup if it is found in two protected stands and so on. The budget constraint is given in (10). The constraint set (11) indicates that the choice variables must be binary.

Given the above-mentioned features, the backup model also takes into account the complementarity of selected sites similarly as the basic model. This model includes, however, a trade-off between protecting a species found in a single stand and protecting a species with backup coverage. It may be optimal to select sites so that many species are found in a single stand without backup coverage or visa versa. Notice also that the backup model does not have any requirement for the primary coverage. This could result in situations where, for example, some species have backup coverage with some species having no coverage at all.

[6] Hogan and ReVelle (1986) originally introduced the idea of backup coverage into location models. Regarding biodiversity conservation, the backup coverage indicates that species will more likely survive when they are present in two protected sites than in one site. We can validate this simply idea, for example, by recognizing that species may go to extinction in a particular site due some catastrophic event, and therefore, the extinction risk is smaller when species are present in several sites. However, species viability obviously depends on several factors, such as population sizes, habitat requirements, dispersal ability, location of protected sites etc. To take into account these factors in site selection requires a good knowledge on the considered species and detailed information on the study area. These requirements are not met typically, as multiple (endangered and rare) species are involved in the reserve selection.

2.3. Data

2.3.1. Study region

The study region is located in south-western Finland (Fig. 1). The study sites lie mainly in the southern boreal zone apart from five sites located in the middle boreal zone or at the border. About 65% of the total area is forestry land. The state owns less than 5% of the forests in the region. Satakunta region has a long history of forestry, and resource extraction started already in the 17th century first for tar production, and later for saw-timber. Extensive cuttings for saw and pulp mills occurred in the 1950s and the 1960s, when the amount of timber extracted exceeded forest growth. Regeneration fellings increased again in the 1990s as forests aged and forest taxation policy changed making cuttings a more desirable option (Korhonen et al., 2000). Practically all forests in the region have been under commercial use, and no genuine old-growth forests are remaining. The ecologically most valuable sites are over-mature semi-natural stands where silvicultural treatments have not taken place for several decades. In southern Finland over-mature forests that have not been treated for 30 years comprise about 4.5% of the productive forest land. Less than 1% of the forest area in Satakunta is protected.

2.3.2. Sites

We included into the study only mature heath (upland) forests, falling into four categories of sites. First, we included sites for which a protection agreement was made (TNV sites). The second category consisted of sites which were offered by the forest owner to the TNV program but which were not negotiated due to the lack of natural values judged during the preliminary field survey. These sites were considered typical managed forests with only a little CWD and other ecological values (managed forest sites, MF). The third category consisted of sites which contained ecological values and were therefore negotiated but for which no agreement was made because of disagreement over the amount of compensation for the 10-year agreement (compensation disagreement, CD). For these three categories, information on site location, their area (ha) and stand volumes were provided by the South-West Finland Forestry Centre. The fourth category included privately owned forests which were not offered to TNV but which were very likely to represent the best as yet unprotected forests in Satakunta region (potential sites, PS). Information on their location and site characteristics were collected in the mid-1990s by the Satakunta Nature Conservation League (Satakunnan luonnonsuojelupiiri). This field survey was a comprehensive and extensive inventory of the remnant semi-natural forest patches in SW Finland. The survey was funded and their data are filed by Satakuntaliitto, an administrative body responsible for developing and coordinating economic activities in the region.

A total of 56 forest stands (sites) were surveyed in this study, with a total area of about 316 ha (Table 1). The surveyed total area and the number of stands are lower in the PS category than in the other categories, because it was difficult to find potential sites for which there was information on stand volumes available (i.e. the site was included in an official forest plan) and for which we got permission for field inventories from the landowners. The size of the sites varied between 1 and 20 hectares (mean 5.6 ha), and there were no significant differences in the average size among categories ($F_{3,\ 52}=0.16$, $p=0.921$). The average stand age was 93 years. The mean stand age and volume were significantly lower in the CD and MF categories than in the TNV and PS categories.

2.3.3. Species sampling

The fieldwork was carried out in June–August, 2004 and 2005. The sites were surveyed by systematic line sampling, with a line width of 8 m. The sampling effort per unit area was held constant by surveying 100 m of line per each hectare of a stand. Thus a 10 ha site, for example, contained 1 km of line and 0.8 ha was surveyed. The first line started 20 m from the nearest forest edge of the sites and subsequent lines parallel 50 m apart. In the case of a very small site, the line was placed as far from the border as possible to minimize possible edge effects. GPS coordinates were taken at both ends of the lines.

Table 1
Characteristics of stands.

	TNV	CD	MF	PS	Total
Number of stands	20	15	15	6	56
Total area, ha	102.6	93.5	85.1	34.9	316.1
Mean stand age, years	102	80	88	107	93
Mean stand volume, m^3/ha	254	163	166	283	209
Species richness					
Total of all stands	59	46	46	31	73
Stand average	14.7	10.3	10.8	12.2	12.2

TNV = trading in natural values agreement sites; CD = sites that were negotiated but no agreement was made; MF = sites that were offered but not negotiated (managed forests); PS = potentially valuable sites that were not offered to TNV.

Epiphytic lichens were studied from the first 50 m of line per each beginning 400 m, i.e., one 400 m^2 sampling plot per 4 ha. On the sites under 4 ha, we placed one sample plot in the middle of the site to minimize the edge effect. All living trees >10 cm DBH within each sampling plot (8 × 50 m) were examined up to the height of 2.5 m for epiphytic lichens. Species taxonomy follows Vitikainen et al. (1997) and Ahti et al. (1999). Lichens were identified in-situ or collected for later identification. In addition, the ten largest *Populus tremula* trunks at each site were examined for a more comprehensive picture of epiphytic lichen species.

Polyporous fungi were examined on the whole transects. Only species that form perennial fruiting bodies, or species which could be detected with the same reliability through the whole survey time were considered. In addition to polypores, two old-growth forest indicator corticoid fungi (*Asterodon ferruginosus* and *Pseudomerulius aureus*) were included into the survey. All visible fruiting bodies of the target species from standing and lying dead wood that were at least 1 m long or 10 cm in their base diameter were tallied. The fungi were identified in-situ or collected for later identification. Tree species, diameter, position and stage of decay were examined from all observed polypores. Species taxonomy of polypores follows Niemelä (2005). Voucher polypore and lichen specimens are preserved in the Finnish Museum of Natural History and the Botanical Museum of the University of Oulu.

2.3.4. Opportunity cost and environmental benefit index

The opportunity cost of protecting a particular stand is the sum of its harvest value and land value in this study.[7] To determine the opportunity costs we first calculated the harvest value of standing timber by using detailed field inventory data on stand characteristics and stumpage prices. The inventory data was provided by the South-West Finland Forestry Centre. Regarding this information, the stands were typically divided into several partitions. Thus, stand values are weighted averages of the partitions where the sizes of partitions were used as weights. Some partitions of stands were not at the cutting age yet. For these partitions of stands we calculated the expected harvest value by multiplying the value of standing timber by the coefficient of expected yield presented in Tapio (2001).

We then assigned the land value for each stand. Land value depends on the particular forest type reflecting its productivity. The

[7] In Finland regional Forest Centres and Environmental Centres often use the additive-value method to assess the value of a forest stand (Oksanen-Peltola, 1994). In this practice the value of standing timber and the land are estimated separately in a similar fashion as in this study. The additive-value method includes also a correction coefficient used to reduce the forest value, because the values of land and standing timber do not take into account administration costs and forest taxes, for example. Determining the value for the correction coefficient is nevertheless a subjective matter, and therefore, we do not use this correction procedure in this study.

stumpage prices and land values used are presented in the Appendix. They are the same as the South-West Finland Forestry Centre has used in the assessment of the conservation value in TNV.

Alternatively, we calculated the opportunity cost of each stand using the Eq. (5). The South-West Finland Forestry Centre provided the information on the actual rental payments, which were paid in one lump sum at the beginning of the contract period in TNV. We used 2% and 4% interest rates in calculations to reveal how the interest rate level affects results.

In the TNV pilot program, the agency (the South-West Finland Forestry Centre) assessed the conservation value of each target, and used this value as a guideline in the negotiations to compare different targets and offers (Gustafsson and Nummi, 2004). The value included prices for different ecological characteristics: large broadleaved trees and pines, dead and burned trees, luxurious vegetation, natural water conditions, distance to existing nature protection areas, size of the area, landscape values and a quick (visual) survey on presence of demanding vascular plants and of particularly species-rich polypore communities. It included also costs of delayed harvesting calculated by using a 1% interest rate for the value of standing timber. An expert from the South-West Finland Forestry Centre made a survey of the forest to gather the required information for the valuation. In this study, we used the values of ecological characteristics of the TNV, MF, and CD stands. We interpreted the sum of the values of ecological characteristics as an environmental benefit index for each stand. Thus, the benefit–cost ratios used in the scoring procedure were constructed by dividing the EBIs by opportunity costs (Eq. (6)).

3. Results

3.1. Number of species

The data from our sampling consisted of 29 wood-inhabiting fungi and 44 lichen species. The total number of species was 73. Altogether 55 species were found in two or more stands and 45 species were found in three or more stands. The stands in the TNV category covered the highest number of inventoried lichen and polypore species among the categories (Table 1). Also the average number of species at the stand level was the highest in the TNV category. Notice that the PS stands included only two species that were not found in any other stand category. Thus, the stands offered to the TNV program covered almost all the surveyed species. The total costs of the field inventory were 40,751 euros.

Protection of all the TNV stands resulted in coverage of 59 species, while the maximum number of species was 73 in the database (Table 1). Because the number of species in the TNV stands was lower than the number of species in all stands, it is obvious that protection of the TNV stands resulted in an inefficient outcome from the viewpoint of maximizing the number of species within the given budget equalling the cost of protecting the TNV stands. To reveal whether this feature remains unchanged or not at lower budget levels, Fig. 2a presents the number of species in the selected conservation network as a function of conservation costs (species–cost relationship). Notice that the slope of species–cost curve reflects the inverse of marginal costs of conservation. The species–cost curves are depicted for cases in which the stands are selected from the set including all the stands, and from the set including only the TNV stands. We do not know the exact order in which the TNV stands were actually selected in the Finnish pilot program, but in Fig. 2 we first selected the TNV stands by maximizing the number of species found in the selected stands subject to the given budget constraint (TNV species). Secondly, we selected the TNV stands in an ascending order of benefit–cost ratio until the given budget was exhausted (TNV scoring). Fig. 2b shows the relative cost differences between cost-effective conservation (benchmark) and the current practice as a function of the number of species found in the conservation network.

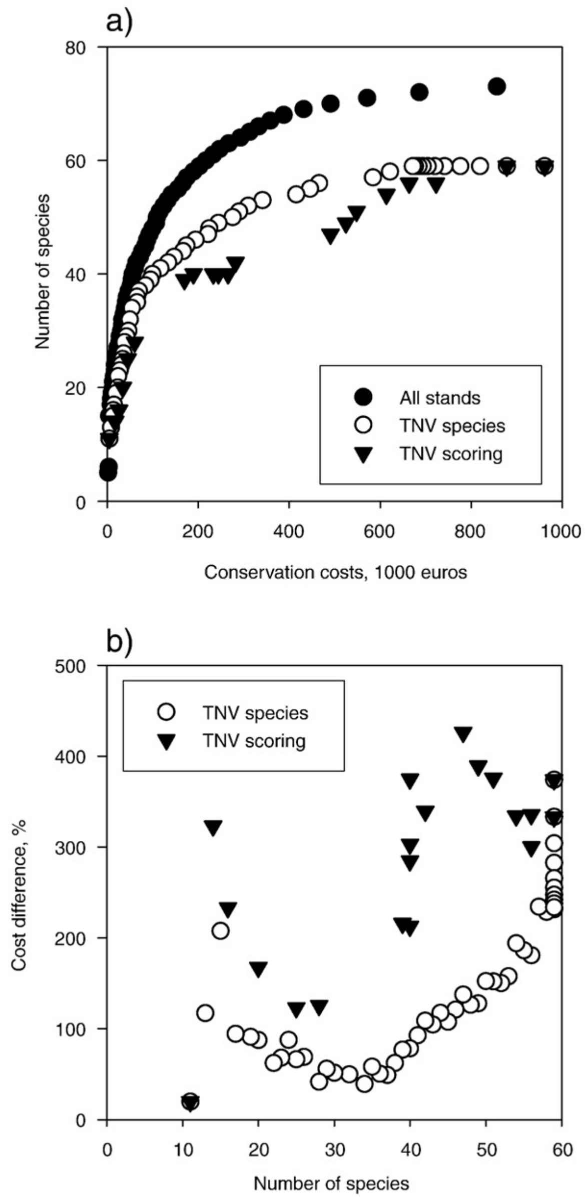

Fig. 2. Species–cost relationships of the alternative site selection strategies (a) and relative cost differences between the strategies (b). "All stands" indicates that all the stands are used in the site selection and "TNV" indicates that only stands belonging to the TNV group are used. The former represents cost-effective conservation and the latter the current practice. The former is used as a benchmark in (b). The TNV selection includes two procedures to rank the stands: maximization of the number of species (TNV species) and scoring by using the benefit–cost ratio (TNV scoring).

To interpret the results of Fig. 2a and b, notice at first that only 19 stands are needed to represent all the 73 species when stands are picked from a set including all 56 stands (see Table A3 in Appendix). Similarly, only ten TNV stands out of 20 available stands are needed to cover the 59 species found in the TNV stands. Therefore, the species–cost curve in TNV species is horizontal after the budget level exceeds 672,000 euros in Fig. 2a, and the cost difference curve in TNV species is vertical when the number of species is 59 in Fig. 2b. Adding the remaining ten stands in the conservation network does not increase the number of species, but increases the costs of conservation.[8]

Fig. 2a shows that species–cost curves are concave, and therefore, the marginal costs of conservation are low when only a few stands are

[8] The remaining stands were selected according to the ascending order of the opportunity costs of the stands in the TNV species selection in Fig. 2.

protected, after which marginal costs start to considerably increase with an increasing conservation level. In particular, it may not be optimal to protect all the species, because the marginal costs of protecting the last uncovered species are very high. This pattern is common in conserving multiple species and does not depend on which set of stands or species is used in the site selection (Ando et al., 1998; Polasky et al., 2001; Juutinen et al., 2004).

The current conservation policy, i.e. protecting the TNV stands, results in a lower number of species than the cost-effective policy at a given budget level or higher conservation costs at the given level of the number of species (Fig. 2a). The relative cost difference between the current policy and the cost-effective policy is large in particular when only a few or a majority of the species are covered (Fig. 2b). When less than 15 species (<20% of total species richness) or more than 45 species (>60%) are covered, the costs of TNV selection are more than two times larger than the costs of the cost-efficient selection. Protecting the TNV stands to cover 59 species costs about 672,000 euros, but using the cost-effective policy the same number of species becomes covered with 203,000 euros. At a low level of number of species, there is obviously much variation in the cost differences, because only one or two stands are protected. Protection of 34 species seems to be an interesting threshold value for the performance of these two policies. At this threshold level, the cost difference between the policies is at smallest (about 40%; neglecting the observations at very low levels of conservation). As the number of species increases or decreases from this threshold level, the cost difference increases.

Given these results, however, it is not straightforward to conclude that the performance of protecting the TNV stands for maximizing the number of species is poor. This is because the results above are a consequence of a simple fact that the total number of species is lower in the TNV species selection than in the case when all four stand categories are included in the site selection (59 vs. 73 species). This feature is largely due to the species–area relationship, a general law in ecology (Rosenzweig, 1995). Indeed, the performance of the current policy is closer to the performance of the cost-effective policy at the lower level of conservation than at the higher level, likely reflecting the fact that the number of species at the stand level is higher in the TNV stands than in the other three stand categories (Table 1).

To assess the robustness of our results we repeated the analysis described above using the opportunity costs that were based on the observed rental payments for the TNV stands. The opportunity costs of the other stands were unchanged in this sensitivity analysis. It turned out that 2.8% interest rate level was the break-even point which resulted in the same opportunity costs of protecting all the TNV stands as when the opportunity costs were calculated according the harvest and land values. However, there were differences between these two cost estimates at stand level. With 4% interest rate the opportunity costs of protecting the TNV stands were 26.1% lower when the rental payment based estimates were used than when the harvest and land value based estimates were used. In contrast, the opportunity costs were 34.4% higher with 2% interest rate, respectively. These figures are small compared with the cost differences between the cost-effective selection (All stands) and the TNV selections (TNV species and TNV scoring) in Fig. 2.

In Fig. 2, the performance of the scoring procedure is the poorest in terms of species conservation. It must be recognized, however, that our comparison does not include transaction costs and field inventory costs (Juutinen and Mönkkönen, 2004). Regarding field inventory costs, the scoring procedure provides some cost savings, because it does not require information on species but information on EBI. The latter is typically easier to gather than the former. According to Gustafsson and Nummi (2004), the salary costs in the TNV trial program were 72 €/ha in 2003. However, only about 40% of salary costs are due to the fieldwork. Accordingly, the inventory costs to estimate environmental benefit indices for the sites included in this study were about 8075 euros. The species inventory costs including fieldwork and species identification were 129 €/ha and the total inventory costs of the TNV species were about 36,275 euros, respectively. Thus, the TNV scoring procedure incurred 28,200 euros less inventory costs than the TNV species selection. This difference can be considered quite marginal (except at the low level of the number of species). For example, achieving the coverage of 47 species in TNV species yielded an opportunity cost of 222,000 euros whereas TNV scoring yielded a cost of 491,000 euros (Fig. 2a).[9]

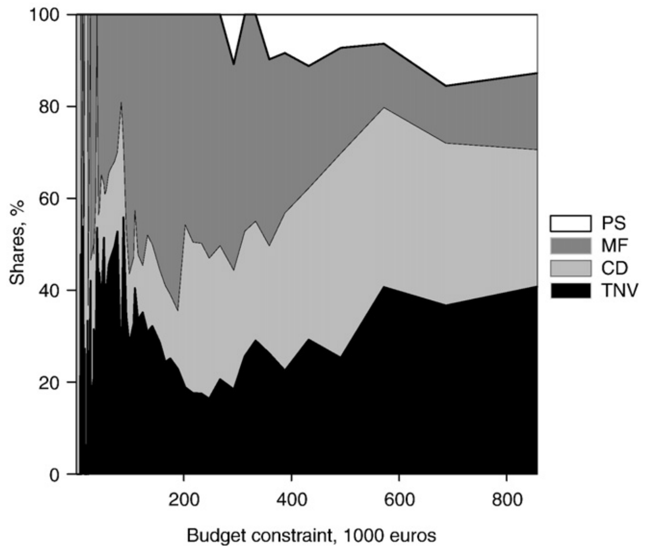

Fig. 3. Proportions of protected stand categories as a function of conservation costs in a conservation network picked by the basic model. The area of each category represents the proportion of hectares of protected stands in that category of the total hectares of protected stands at different budget levels.

In any case, the results show that some species may not be covered under the current policy and the site selection is biased from this perspective. Therefore, it is interesting to consider next how this bias appears in terms of the way in which the different stand categories are represented in a cost-effective conservation network.

3.2. Selected categories

Recall, our data includes four stand categories, and the TNV stands comprise one third of the total area under consideration. If protection of the TNV stands were a cost-effective policy, the TNV stands should be disproportionately well represented in the selected network ('33% of the total area) picked by the basic model. Fig. 3 shows proportions of areas of these categories as a function of conservation costs in a cost-effective conservation network.

The TNV stands do not seem to have a dominating role in maximizing the number of all species in the selected conservation network, as many protected stands do not belong to this category (Fig. 3). On the average, the share of the TNV stands of all protected stands is 29%. It is clear that the share of the TNV stands is less than 100% at the budget level exceeding 672,000 euros, because at a maximum only 59 species out of 73 can be covered by protecting the TNV stands. At lower budget levels, however, the share of the TNV stands could technically be 100%, but it is much lower. Interestingly, the MF stands (i.e. typical commercial forests) have a large share in a cost-effective conservation network, in particular, at a low cost level. This reflects the fact that the opportunity costs of the MF stands are

[9] Similarly, we can take into account the inventory costs while comparing the costs between the cost-effective selection and the TNV species selection in Fig. 2a. Recall, the inventory costs of the cost-effective selection were 40,751 euros. Thus, the cost saving in the inventory costs is only 4476 euros. In contrast, the cost saving is 32,676 euros in comparing the cost-effective selection and the TNV scoring selection.

Table 2
Selected stands picked by backup of two stands.

	TNV	CD	MF	PS	Total
Number of protected stands	10	8	5	1	24
Protected area, ha	62.4	41.2	29.4	5.9	138.9
Conservation costs, euros	453,086	149,384	114,136	140,159	856,765
Number of species	56	40	36	14	72
Backup of two stands*	47	34	32	13	52

*Number of species with backup coverage of two stands.

typically lower than the opportunity cost of the other stands. Previous studies have showed that at low budget levels it is optimal to select stands having the lowest opportunity costs (Juutinen et al., 2004). In contrast, the PS stands are selected only at high budget levels. The opportunity costs of these stands are the highest on average. The cost-effective conservation network also includes many CD stands. Indeed, the average share of these stands (36%) is the highest. The average opportunity costs of the CD stands are the second lowest.

The above results reveal many important features of species conservation. First, if all the species are considered as important for conservation, typical commercial forests (MF stands) have a noticeable role in establishing the network of reserves. Thus, these stands should not be neglected when designing a reserve system. Commercial forests, however, do not represent a threatened habitat type, and therefore, these forests do not require strict protection.

Second, from the viewpoint of maximizing the number of species in the selected conservation network, the current voluntary conservation policy has not managed to protect the stands cost-effectively. In particular, a cost-effective conservation network would include many CD stands. This outcome indicates that the agency of the Finnish pilot program has likely offered too low compensation for protecting these stands as the landowners did not participate in the program at the offered level of compensation.

Finally, a voluntary incentive mechanism does not capture all ecologically valuable targets (a cost-effective conservation network includes also PS stands), and therefore, some species remain uncovered in the TNV program. However, a cost-effective conservation network does not include all the PS stands but only some of them. In this sense, the voluntary conservation has not caused a systematic bias in the site selection.[10] The presence of some uncovered species is a result of the use of a scoring procedure in the pilot program that does not reflect effectively the complementarity of protected areas. It is interesting to investigate next whether this outcome holds or not when species persistence along with the complementarity is taken into account in the site selection.

3.3. Backup coverage

In this sub-section, we consider a backup coverage of species (a species is covered in two or more selected stands) with the budget that equals to the cost of conserving all the TNV stands (962,000 euros). Thus, all the TNV stands could be selected in a cost-effective conservation network as was done in the pilot program. Let us first assume that the environmental agency considers only species found in a single stand and species found in two stands when selecting the conservation network (backup of two stands). Then we extend the analysis to the backup of three stands. Notice that all 73 species can be covered (without considering the backup coverage) given this particular budget (Appendix, Table A3).

The results of having a backup of two stands for protecting all species are presented in Table 2. In this case, the total number of

[10] Recall, the PS stands include only two species that are not found in any other stand category.

Table 3
Selected stands picked by backup of three stands.

	TNV	CD	MF	PS	Total
Number of protected stands	14	8	5	2	29
Protected area, ha	68.5	41.2	22.9	7.9	140.5
Conservation costs, euros	594,395	183,885	136,791	45,474	960,545
Number of species	56	40	35	21	72
Backup of two stands*	46	34	31	20	51
Backup of three stands**	33	22	22	12	42

*Number of species with backup of two stands, **number of species with backup of three stands.

protected species is 72, and 52 species have a backup of two stands.[11] The number of species found in two or more stands is 40 if the conservation network is selected to maximize only the number of species (Table A3). Thus, the conservation network presented in Table 2 is a compromise between single and multiple representations of species. In other words, there is a trade-off between species richness and persistence in this site selection strategy. However, the backup coverage can be guaranteed for large numbers of species with little reduction in primary coverage as was shown by Malcolm and ReVelle (2005). Notice also that the conservation costs in Table 2 are somewhat lower than the available budget (962,000 euros), because in a discrete model, there may not be optimal solutions that meet exactly the given constraint. Species found in the MF and PS stands have the highest relative backup levels, i.e. about 90% of the species reside in two or more stands. However, stands providing these backup occurrences mainly represent the TNV or CD stand groups. In contrast, the backup occurrences for species found in the TNV and CD stands originate mainly from stands belonging to the same stand category.

Let us next assume that the agency takes into account species found in a single stand, species found in two stands, and species found in three stands when selecting the conservation network. The results of backup of three stands are presented in Table 3.

In the case of a backup of three stands, the total number of protected species is 72, 51 species have backup coverage of two stands, and 42 species have a backup coverage of three stands (Table 3). The figures are not at a maximum. For example, the maximum number of species having a backup of three or more stands is 44 when the conservation network is selected to maximize only this conservation aspect subject to the given budget. The species found in the MF (89% of species have a backup presentation) and the PS stands (95%) have the highest backup levels of two stands, and TNV stands the lowest (82%). However, backup occurrences for species found in the TNV stands stem largely (83% of the backup occurrence) from the other TNV stands. In contrast, respective figures of internal backup presentations are less than 59% for species found in the CD, MF, and PS stands. Considering the backup of three stands, the results follow a similar pattern.

The results of backup models (Tables 2–3) show that the share of the TNV stands in the selected conservation network increases when species persistence is considered in the site selection along with species richness (>44%), compared with the case that only species richness is considered (29% on average; Fig. 3). In addition, the share of the TNV stands is larger in the backup of three stands (49%) than in the backup of two stands (45%). Thus, taking into account species persistence together with species richness in the site selection, the results to some extent emphasize the utility of protecting the TNV stands. It can be seen, however, that also in the backup approach a cost-effective conservation network includes stands belonging to the CD, MF, and PS categories. This outcome reveals that the current

[11] These figures are not at the maximum in a sense that they could be higher if species richness and backup coverage are maximized separately under the given budget. The maximum figures under the given budget are 73 and 54, respectively. If the conservation network is selected to maximize only the number of species having a backup with two stands, for example, the number of species in this network is 67.

voluntary policy results in biased site selection, and therefore, its performance is not optimal.

4. Discussion and conclusions

Conservation contracting programs have potential benefits over the traditional conservation actions where conservation authorities govern, because of wide social acceptance and flexibility. Such programs may also result in ecologically justifiable network of protected sites if site selection procedures are appropriate (Mönkkönen et al., 2009). However, for voluntary programs to be efficient, a large pool of bids is needed to ensure competition among bidders. Otherwise, there is a risk of severely inefficient outcome as landowners may shade their bids above their opportunity costs (Juutinen et al., 2008a).[12] The number of potential participants must be large also from an ecological viewpoint so that the complementarity approach can effectively be used.

Our results suggest that the participation in the TNV pilot program has been large enough to meet the ecological goals in a sense that the cost-effective conservation network included only a few targets that were not offered in the pilot program. This result supports the finding that the landowners in the region were properly informed about the program (Juutinen et al., 2005). In addition, the program was found attractive among landowners, and therefore, contract features, such as the length of the contract period, were likely to be suitable for the forest owners (Horne, 2006). However, in a voluntary program compensation payments must be large enough to cover the opportunity costs (Latacz-Lohmann and Van der Hamsvoort, 1997). Our results suggest that the agency of the Finnish pilot program may have offered, at least partly, too low compensation payments as the cost-effective conservation network included many sites that were offered, but for which no agreement was made because of disagreement over the amount of compensation. Thus, the contract allocation method used in the Finnish pilot program could be improved.

Along with the number of participants, the representativeness of a selected conservation network in a voluntary program depends on the measurement of the ecological quality of the offered targets. Connor et al. (2008) argued that there are potentially very large returns associated with the improved environmental targeting capacity due to conservation auctions. The argument considered conservation programs of agricultural land, which applied uniform payment policy and in which the targets were selected on an “as they arise” basis without the advantage of prioritising bids based on knowledge of the environmental benefits provided. In forest conservation programs, in particular, it is likely justified to use a discriminatory pricing policy as the opportunity costs vary strongly among targets (Stoneham et al., 2003; Ferraro, 2008). Similarly, the ecological characteristics vary, and therefore, an accurate quality assessment is needed to prioritise the targets.

Our results suggest that the use of scoring procedure, for example ranking targets using stand level measurement by environmental benefit index per bids as an allocation rule, is not a cost-effective mean to allocate contracts. One can argue that the two methods, scoring and detailed species inventory, are not measuring the same parameters. However, most of the used scoring parameters (large broadleaved trees and pines, dead and burned trees, luxurious vegetation, and a quick survey on demanding vascular plants and especially species-rich polypore communities) can be considered as surrogates for total species richness and richness of rare and threatened species. Some of the parameters, such as natural water conditions, distance to existing nature protection areas, size of the area and landscape values refer to the naturalness of the site or landscape-level parameters, which may turn important for the long-term persistence of species. However, these surrogates matter only if the species exist in the studied stands. Furthermore, because all the sites in our evaluation were mature heath (upland) forests, the scoring method did not reveal differences in habitat types. Comparing these two methods can be justified also as a quality assessment of the pilot programme: the scoring method was a practical tool used for quick assessment of the natural values of the offered targets, whereas the species inventory was used a tool to assess the quality of the used method.

Even though the scoring procedure was apparently not sufficiently effective to give detailed information on the overall diversity of forest-dwelling species for ranking the sites cost-effectively, it seemed to be ecologically relatively effective to reveal many of the sites hosting rare and threatened species, as the number of those species was generally higher in the TVN stands compared with the other categories (Mönkkönen et al., 2009). We did not apply any model to give more weight to the rare or threatened species partly because such weights are inevitably subjective (Juutinen and Mönkkönen, 2007). The results in Mönkkönen et al. (2009), however, indicate that giving more weight to red-listed species would have emphasized the TNV stands in a cost-effective network.

The scoring procedure may also be appropriate if species persistence is strongly emphasized in the contract allocation as it was tilted in favour of the TNV stands. Our result that second and third representations of the species found in the TNV stands originated predominantly from the other TNV stands suggests that species composition in the TNV stands differed from species composition in the other stand categories. Therefore, the TNV stands are important from the species persistence perspective. However, the conclusion that the scoring procedure yields population persistence may be idiosyncratic for our data and the performance of scoring procedure should be tested against species viability data to check for the generality of our finding.

Scoring is a typical site selection tool in voluntary programs because time and funding are limited to carry out comprehensive species inventories. Our results suggest that it could be justified to use, for example, information on indicator species in allocating the contracts, so that the representativeness of protected areas is taken into account in site selection (Juutinen and Mönkkönen, 2004). Surveying indicator species causes some extra costs, but these may be minor compared with the costs savings in opportunity costs due to the improved targeting of protected areas (Wikberg et al., 2009). However, the selection of the indicator species strongly affects the inventory costs. The species groups which demand highly specialized professionals to identify the species may be costly, e.g., the inventory costs of beetles in boreal forests were 7–10-fold compared with the inventory costs of birds, vascular plants and polypores (Juutinen et al., 2006). If the species data are not available, one useful proxy would be using an environmental diversity approach in which it is also possible to take into account the stand interdependence in biodiversity assessment (Faith and Walker, 1996; Faith et al., 2003; Juutinen et al., 2008b).

It is important to notice that we did not consider the location of protected areas in this study. Regarding species persistence, protected areas should be located near each other so that species could move from an area to another. Thus, also from this viewpoint, it is important to have a large pool of bids.[13] In Southern Finland, however, it is difficult to establish continuous conservation networks, because the forest ownership is fragmented with small stand size. One approach to promote offers of neighbouring targets is to pay a bonus for

[12] Notice that using quasi markets to allocate conservation contracts does not remove totally the information asymmetry between landowners and the environmental agency, and therefore, landowners will always shade their bids to some extend in voluntary conservation programs (Latacz-Lohmann and Van der Hamsvoort, 1997).

[13] Focusing on southern Bahia in the Brazilian Atlantic Forest, Chomitz et al. (2006) showed that a voluntary incentive system resulted in a reserve network characterized by large, viable patches of contiguous forest without explicitly planning this outcome, provided forest cover is spatially autocorrelated. The finding is based, however, on simulation, not on an actual conservation program.

aggregation of the targets (Parkhurst et al., 2002). Another important issue is that there may not be many targets with very high ecological quality in the study region (Juutinen et al., 2008c). Therefore, conservation contracts should be flexible, including alternative possibilities for landowners to provide biodiversity in their lands. In particular, it could be efficient to restore the ecological values of the low cost sites.

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Appendix A

Table A1
Stumpage prices, €/m^3.

Tree species	Saw timber	Pulpwood
Pine	47.10	15.00
Spruce	43.80	23.80
Birch	37.90	13.50
Other	33.60	13.50

Table A2
Land values.

Habitat type	€/ha
Grovelike	639
Fresh	404
Dryish	336
Dry	269

Table A3
Selected stands in protecting all 73 species by using the basic model.

	TNV	CD	MF	PS	Total
Number of protected stands	7	6	4	2	19
Protected area, ha	50.9	37.2	20.8	15.9	124.8
Conservation costs, euros	453,086	149,384	114,136	140,159	856,765

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