
Volume 11, issue 7, November 2008
ISSN 1462-9011
ELSEVIER
Environmental Science & Policy
Editor-in-Chief
James Briden

ENVIRONMENTAL SCIENCE & POLICY 11 (2008) 613–626

available at www.sciencedirect.com

ScienceDirect

journal homepage: www.elsevier.com/locate/envsci

Cost-effective forest conservation and criteria for potential conservation targets: a Finnish case study

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ARTICLE INFO

Published on line 1 July 2008

Keywords:
Biodiversity services
Conservation planning
Finland
Forest resources
Habitat quality
Remote sensing

ABSTRACT

Selecting reserves for forest biodiversity maintenance is often done by setting criteria for components of structural elements of biodiversity, such as a volume of decaying wood. We tested how the different threshold values for the components of structural elements affect the cost-effective site selection. Using Finnish National Forest Inventory information and remote sensing data, we determined a habitat quality index and economic value for each site in Satakunta region in Finland. Moreover, we defined several sets of potential conservation targets using alternative criteria for the habitat quality index developed for the Finnish case study. These figures were used in the site selection model in order to maximize the sum of habitat index of selected areas under a given budget constraint. We found that the production possibility frontier for the outputs of timber and biodiversity is only slightly concave when using the given threshold values. Thus, the optimal combination of the outputs is sensitive to the relative values of these goods. Our results suggest that an integrated approach in forest conservation could provide to environmental managers considerable cost savings compared with current management practices. Environmental managers could also reduce conservation costs by loosening the criteria for potential conservation targets. This would not lower considerably the quality of conserved forests.

1. Introduction

Economic forces drive much of the extinction of the world's biological resources and biodiversity (Pimm et al., 1995; Vitousek et al., 1997). In particular, human interventions have increased wood production and this has resulted in degradation of forest biodiversity and ecosystem services (Millennium Ecosystem Assessment, 2005). Thus, modifications of ecosystems to enhance one service generally have come at a cost to other services due to trade-offs. However, the objectives set for forest use are nowadays more diverse than in the past. In this sense non-timber values, such as recreational activities, forest carbon sequestration, maintenance of biodiversity, microclimate, protection of erosion and water regulation, have gained prominence alongside traditional wood-production values (Costanza et al., 1997, 1998). Therefore, it is important to have methods for allocating limited resources efficiently to alternative uses and to develop procedures to identify the parcels of land where conservation efforts should be directed.

Recognising the links between biodiversity and ecosystem services would help stakeholders to avoid biodiversity losses that lead to unacceptable losses of ecosystem services. Due to

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1462-9011/$ – see front matter
doi:10.1016/j.envsci.2008.05.004

the trade-offs between ecosystem services and their complex relationship we emphasize that is important to conserve and make a sustainable use of forest resources over large areas or reserve networks (Gutzwiller, 2002; Miller and Lanou, 1995; Soulé and Sanjayan, 1998; ReVelle et al., 2002; Rodrigues and Gaston, 2002). In addition, decisions on the use of natural resources should be based on a comparison of the expected monetary value of the harvested products and the values associated with the ecosystem goods and services foregone because of harvesting (i.e. an integrated approach considering both economic and ecological values for site selection). This means that in practice areas are selected into a conservation network according to their benefit–cost ratio (Weitzman, 1998). Thus, it should be a compromise between the sites that provide high benefits in terms of biodiversity services and those with a reasonable cost. These features can be captured into the decision-making by using numeric optimization tools, such as site selection models, in conservation planning.

Adequate selection of nature reserves for biodiversity conservation has been under extensive research over the past two decades (Cabeza and Moilanen, 2001; ReVelle et al., 2002; Moilanen and Cabeza, 2002; Rodrigues and Gaston, 2002; Mikusinki et al., 2007). Even if site selection models have improved in the last two decades, their impact in applied conservation planning remains minimal (Cabeza and Moilanen, 2001; Martin-Lopez et al., 2007). Typically, previous integrated studies have focused on species conservation (Ando et al., 1998; Calkin et al., 2002; Nalle et al., 2004; Millennium Ecosystem Assessment, 2005). For instance, this approach is appropriate when the aim is to illustrate how to apply developed methods. Nevertheless, because of the cost and time involved to obtain information on species, for practical decision-making more cost-effective way to measure biodiversity is needed (Juutinen and Mönkkönen, 2004; Moilanen and Wintle, 2006).

One attempt to solve this problem is to use variation in structural elements as proxy for species composition (Faith and Walker, 1996; Noss, 1999; Lindenmayer et al., 2000). Defining specific criteria or threshold values for these components or variables within the system becomes a crucial step in environmental management. In fact, these thresholds in criteria determine the pool of potential target sites, i.e. the size of production possibility set. The more stringent the criteria are the smaller is the pool of potential targets. The criteria also have an effect on the benefit–cost ratios of these targets. This may have severe impacts on what is being protected when the conservation budget is limited and all potential targets are not protected.

The aim of our study was to find out how the different threshold values for components of structural forest elements affect the cost-effective site selection. The work focuses on the Satakunta region, south-west of Finland. The long history of forest monitoring and management in Finland allowed the analysis of cost-effective forest conservation from several points of view. First, we considered the threshold values in terms of production possibility frontier (PPF) revealing whether or not the trade-off between biodiversity and timber production has a similar pattern under alternative threshold values. We also compared the approximated current management practice with the most efficient management scheme described by PPF. Next, we investigated how the costs of conservation are affected by the threshold values. Finally, we considered how the ecological quality of selected sites varies when different threshold values are used.

In this paper, we focus on structural elements of biodiversity. There exists previous studies on this issue (Kangas and Pukkala, 1996; Siitonen et al., 2002), but it has not been systematically analysed how different threshold values for components of forest structural elements affect cost-effective site selection. Thus, an explicit aim of our contribution is to address some straightforward practical policy implications. In addition, to emphasize applicability we develop a method for site selection using habitat quality evaluation based on existing Finnish National Forest Inventory (NFI) data in combination to remote sensing data.

We emphasize the efficient use and added value of existing data by introducing an operational habitat index to assess the status of forest protection and ecologically valuable habitats. Thus, this work explores a rapid biodiversity assessment method using spatial analysis of existing remotely sensed data. Although this approach is not new, it has rarely been applied to simultaneously take into account both ecological and economic aspects of forests. We use term "habitat quality" instead of "habitat suitability" (Store and Kangas, 2001) to emphasize that we do not focus on any particular species and its habitat requirement. In contrast, we apply the habitat index to distinguish high-quality targets from low quality targets from the pool of potential target sites representing semi-natural old-growth forests with large amount of decaying wood (coarse woody debris, CWD). The habitat index used in this study is based on habitat features such as the volume of decaying wood and the level of human impact (naturalness). Many rare and threatened species are dependent on these habitat features (Bader et al., 1995; Siitonen, 2001; Dettki and Esseen, 2003) and they are considered important guidelines for environmental regulators in Finland (Ympäristöministeriö, 2003).

Within this context, our analysis provides useful information for environmental managers to set appropriate criteria for potential conservation targets and use limited resources cost-effectively. However, given the limited accuracy of remote sensing data our results can serve only as a starting point (early stage planning). Field level evaluation will be advised for a more detailed level conservation planning known as final-stage land-use planning (Hilli and Kuitunen, 2005). Nevertheless, given high-level commitments to reducing the rate of biodiversity loss by 2010, there is a pressing need to develop simple and practical indicators to monitor progress (Rouget et al., 2006; Faith et al., 2008). In this context, a biodiversity quality index is proposed, in tandem with cost-effect analysis to provide an overall indicator suitable for policy makers and decisions on protection of the most appropriate areas from a biodiversity point of view.

2. Methods and materials

2.1. Study area

Our study focuses on forest land in south-western Finland (Fig. 1). More precisely, we focus on forestry land that

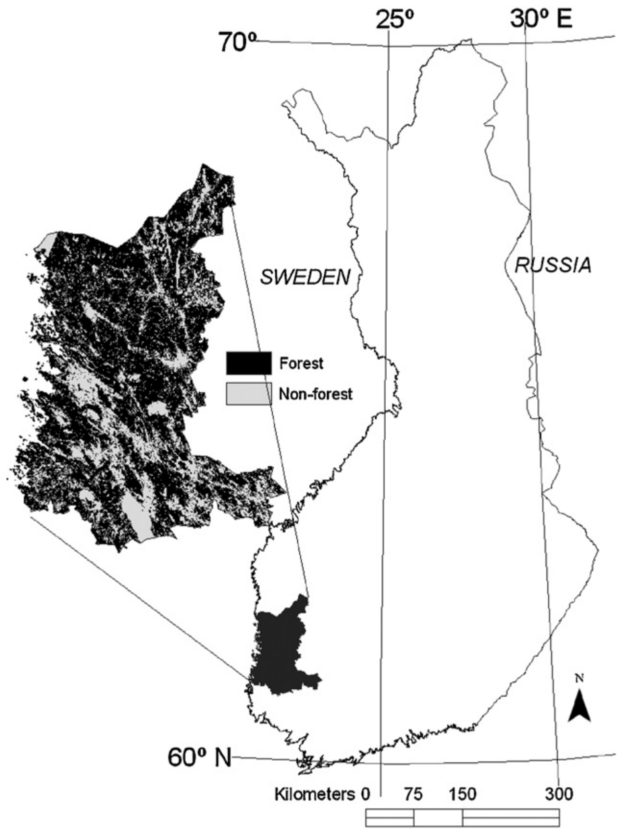

Fig. 1 – Map of Finland and the study area, Satakunta.

comprises forest land, poorly productive forest land and unproductive forest land. Protected forests include national parks, strict nature reserves, wilderness areas, old forest protection areas, areas in the old forest conservation program, peat land protection areas, herb-rich forest protection areas, privately protected areas, special protected areas and areas protected on the decision of the Finnish forest and park service. Accordingly, the share of the protected forests in the entire country is 8.9% of the combined forest and poorly productive forest land on the basis of the national definition (Finnish Forest Research Institute, 2006). The corresponding proportion for southern part of Finland is 2.2% and Northern Finland 15.8%. A higher proportion of forestry land is privately owned in south Finland than in North Finland (74.3% in the southern part of the country and 39.3% in the north; Finnish Forest Research Institute, 2005) making forest protection more complicated in the south. The forests are also more productive in the south than in the north, wherefore the most part of the annual cut comes from southern Finland. In southern Finland, there is a recognised lack of conservation areas with high ecological quality (e.g. Virkkala et al., 2000). Considerable pressures exist to enlarge the network of protected areas in southern Finland, because most of the endanger species and/or species at risk are also concentrated to the southern part of the country (Hanski, 2000). Forestry and associated changes in the structure and configuration of forest stands and landscape level have resulted in the loss of biodiversity in all Fennoscandian countries (Esseen et al., 1997). Thus, the critical question is in which terms and how to invest a limited budget to conservation purposes in a cost-efficient way.

We selected Satakunta region, SW Finland, in order to test our approach (Fig. 1). The region belongs to south boreal forest vegetation zone but small parts of it are in hemiboreal and middle boreal zones. The land area of Satakunta is some 830,000 ha including 595,000 ha forestry land (combined forest land, poorly productive forest land and unproductive forest land). Protected areas currently cover about 2% of forestry land (9285 ha) in the study area. In conservation literature, a ten percent goal has often been mentioned as being the target value in terms of the proportion of protected areas of the total land area (Angelstam and Anderson, 2001; Rassi et al., 2001; Hanski, 2003). We use this as a reference point in evaluating our results. We further consider 5% protection level as the intermediate reference point between the current situation and the suggested target of 10%.

2.2. *Finnish National Forest Inventory*

All spatial data used in this study are derived from the Finnish NFI and the multi-source Finnish National Forest Inventory (MS-NFI). The Finnish NFI has been producing large-scale information on Finnish forests since the 1920s, and forests statistics for small areas have been computed since 1990 using satellite images and digital map data in addition to field measurements by means of MS-NFI (Tomppo, 2006a,b). It is extremely important for conservation purposes to develop methods that can use existing data because collecting data from large areas is time and resource consuming, and using existing data likely saves limited funds for the conservation action. This is particularly the case when the data-source fulfils high-quality standards and avoids error propagation that is common in many multi-source large-scale data (Burrough and McDonnell, 1998). We used the pixel level predictions of selected forest variables as input data for the models in this study in addition to interpolation layers calculated for some NFI field plot data from the 9th rotation of the NFI (in years 1996–2003).

The sampling unit used in the 9th rotation of the NFI is a field plot. Field plots are arranged to clusters to make the design as cost-efficient as possible. The sampling design has been adapted to the variability of the forests resulting to 67,264 plots on forestry land in all of Finland with varying field plot size from South to North Finland (Tomppo, 2006a). Different plot sizes have been applied for different variables, e.g. angle gauge plot field tree measurements, with a maximum radius of 12.52 m in the South (basal area factor of 2) and 12.45 m in the North (basal area factor of 1.5), and 7 m for dead wood and 30 m for key habitats. Pixel level predictions in map format were produced in the multi-source Finnish National Inventory (MS-NFI) based on *k*-nearest-neighbour (*k*-nn) estimation and its improved version (Tomppo, 1991, 2006b; Tomppo and Halme, 2004). MS-NFI procedure assigns field plot data of forest inventory from the nearest field plots (in the image featured space) to all satellite image pixels (Tomppo, 1991; Tomppo and Halme, 2004). Digital maps are used to delineate forestry land from other land-use classes. An essential property of this method is that all inventory variables, typically 150 measured in the field, can be predicted for all pixels and forest parameter estimates (e.g. mean volumes of growing stock by tree species) derived at the same time for the computation units. Another advantage is that area statistics and thematic maps are produced by the same method (Tomppo, 2006b).

It must be mention that one of the problems encountered when working with MS-NFI, is the complexity on the error estimation of the estimates for an arbitrary area. We use here the empirical error estimates for MS-NFI as it was calculated for the 9th rotation of the NFI summarised in Tomppo et al. (2008a, 2008b). Relative root mean square errors (RMSEs) of 5, 12, 15 and 16% for mean volume and mean volumes of pine, spruce and birch, respectively, were obtained in seven test units of size 100 km^2 when the k-nn estimates from the 9th rotation of the NFI. The relative RMSEs decreased to 4% when the area increased to 1000 km^2. For more detail on the sources of error regarding the original data used in this study refer to Tomppo et al. (2008b).

2.3. Habitat index

2.3.1. Variables and their classification

Locating habitats that have particular attributes for protection and analysing their distribution has been the primary goal in constructing the habitat quality model resulting in a habitat index. Habitat quality assessment and habitat suitability maps could be useful in regional management planning, e.g. when extending the existing protected areas network for protecting certain species or habitats of particular importance in managed forest (Rautjärvi et al., 2004; Luque and Vainikainen, 2006). In Finland, a high proportion of commercial forests is within the sphere of woodlot-specific forest planning, which enables the use of a forest decision support system as a link between ecological knowledge and practical forestry (Store and Jokimäki, 2003; Nuutinen et al., 2001). In order to improve this existing forest management planning, it is essential that the decision alternatives be assessed with respect to a combination of expert knowledge and habitat models. A challenge is to develop methods and practices of locating and evaluating suitable sites for threatened species. The problem is that in the case of biodiversity conservation empirical evaluation models based on real field data for all species of interest cannot be expected to become available. One way of dealing with this problem, as proposed in this work, is to use habitat quality indices that reflect the quality of the forest by identifying possible causal relationships between forest structure, environmental data, and ecological conditions. We depart from the hypothesis that all species have specific habitat requirements, which can be described by habitat factors. These factors are connected to the critical characteristics of the habitat, e.g. to those of vegetation or soil, but also areas surrounding the habitat can influence the habitat quality (e.g. spatial structure of landscape elements). Within the framework of this study, the habitat index reflects the value and importance that an area potentially possesses in terms of biodiversity. The habitat index was used as the sole ecological variable in optimizing the site selection for conservation. The first step in assessing the quality is to determine the forest habitat factors on the basis of an analysis of existing studies and knowledge. Here, judgements made by experts on ecology were applied, in particular key species requirements in terms of forest habitat suitability (Virkkala, 1996; Väisänen and Järvinen, 1996; Hildén et al., 2005; Romero-Calcerrada and Luque, 2006).

The model to produce the habitat index is a simple additive approach based on forest structure characteristics derived from the NFI data and remote sensing. The inputs in the model were predicted volume of growing stock (from MS-NFI), predicted stand age (from MS-NFI) and predicted productivity of the site (from MS-NFI), as well as volume of dead wood as defined in NFI, also called coarse woody debris (CWD) here. All input data for the model are from MS-NFI thematic maps and NFI plot level data. The original resolution of 25 m of MS-NFI maps was shifted to a 200-m resolution in order to facilitate the model calculations and as well to reduce the effect of prediction errors in the calculations. In this way, the Satakunta region was partitioned into 148,812 raster cells, which we call sites. Each site has an area equal to 4 ha, which is near a typical stand size within this study region. Each cell is assigned a habitat index derived from the model. Note that these artificial operative units may include, for instance, arable land and watercourses along with forestry land. Thus, area of forestry land may be lower than 4 ha and differ among sites.

Volume of growing stock and stand age are basic variables depicting forest structure. Many threatened and rare species prefer old stands with high volume of growing stock (Esseen et al., 1992; Kuusinen, 1995; Dettki and Esseen, 2003; Berglund and Jonsson, 2005). Volume of growing stock is the stem volume of all living trees above stump height (with a minimum height of 1.3 m) derived from field plot level measurements, and predicted for pixels (m^3/ha). Stand age is the weighted mean age of the trees of the main tree storey, the volume of a tree as the weight.

The thresholds for reclassifying the values of these variables were chosen to balance the following aspects: the ecological grounds for the reclassification, the patterns that can be observed and the restrictions the multi-source method puts upon the variables. The output maps were recoded after shifting the resolution (Table 1).

The thresholds for the five volume classes were chosen carefully to show patterns of difference through southern and central Finland. Mature forests in the region typically contain more timber than 240 m^3/ha (class 5) and other classes can be considered representing clear-cut or sapling stands (class 1), young thinning stand (class 2), advanced thinning stand (classes 3 and 4). As naturally originated early successional stages are very rare in Finland we assume that sites with low timber volume represent lower habitat quality sites than sites with high timber volume. Using similar logic forest age was classified in four classes with increasing habitat quality from sapling and young thinning stands (class 1) to clearly over-mature or old-growth forests (class 4). Class 2 represents advanced thinning stands and mature forests (typical forest rotation in southern Finland is 70–80 years), and class 3 mature to over-mature forests.

The productivity of a site has been shown to be a key determinant of species richness of a site (Mittelbach et al., 2001). In NFI productivity is the average increment of the growing stock of the corresponding site fertility class over a

Table 1 – Input layers (after reclassification)

Code	VOL (m^3/ha)	AGE (years)	BON (m^3/(ha a))	CWD (m^3/ha)	NAT (sites/km^2)
1	0–40	0–40	0–2	<0	0–0.072
2	41–80	41–80	2.1–3	0–2.5	0.073–0.108
3	81–160	81–120	3.1–4	2.51–5	>0.108
4	161–240	>120	>4	5.01–10	
5	>240			>10	

VOL, volume of growing stock; AGE, stand age; BON, productivity of the site; CWD, volume of coarse woody debris above regional average; NAT, degree of naturalness.

forestry centre also termed bonity. Bonity is independent of growing stock and forests age as it reflects the potential average increment of a site type class. The same classification of bonity was employed as earlier in forest income taxation. For MS-NFI estimation, the average increment of the volume of growing stock was computed on the basis field data of NFI8 by site classes and by forestry centre. These average increments were attached to site classes of field data and predicted for each pixel using *k*-nn estimation (Tomppo and Halme, 2004).

The volume per hectare of coarse woody debris is considered one of the key attributes in boreal forests with a high biodiversity value (e.g. Esseen et al., 1997) and there is a great difference in the volume of CWD between managed forests and natural or semi-natural forests (Martikainen et al., 2000; Siitonen, 2001). The volume, quality and roughness of CWD are also measured in the Finnish NFI, and for the southern and central Finland we have some 53,000 field plots in use for producing the CWD layer. CWD in NFI is defined as pieces of dead wood with a minimum length of 1.3 m and with a minimum diameter of 10 cm. For each NFI plot, the average volume of CWD per hectare has been calculated by tree species and separately for standing and lying and by diameter classes and decay classes (Tomppo et al., 1997). For each 4 ha grid cell, we derived total CWD volume (m^3/ha) from the field plot data using the Kriging method in ArcGIS Geostatistical Analyst (Johnston et al., 2001). In the habitat quality model, the value for the volume of CWD is expressed through a difference between the estimated grid cell volume and the regional average volume, the average volume being 1.82 m^3/ha for the Satakunta region. This was done to take into account the regional differences in the volume of CWD in southern and central Finland (the original extent of the habitat quality model); the regions used are forestry centres (governmental districts), and the average CWD volume ranged between 1.18 and 4.50 m^3/ha between the forestry centres.

The thresholds for the five classes of the CWD layer were defined as follows. The NFI field plot with high CWD volumes (above 15 m^3/ha) were quite sparse outside the protected areas as only about 5% of all plots had a CWD volume of at least 15 m^3/ha. Therefore, the highest class (class 5) was defined to include sites with the CWD volume of at least 10 m^3/ha above the regional average. This is well above the average CWD values for protected areas in southern Finland (7.5 m^3/ha; Tonteri and Siitonen, 2001). Class 4 represent CWD values typical for south Finnish protected areas. CWD values in classes 2 and 3 are above regional average but below those in south Finnish protected areas, and class 1 include sites with CWD less than regional average.

In addition, silvicultural history has also been evaluated on all plots on forestry land. The variable noted as ''naturalness'', is used as a surrogate for ''non-managed forest''. Non-managed forests in Europe are rare and many rare and threatened species are dependent on long continuity of habitat features such as dead wood or large living trees. Therefore, silvicultural history, i.e. previous fellings, soil preparation and other silvicultural measures with their dates, provides information that is not necessarily captured by present site features such as CWD or volume of growing stock (see Penttilä et al., 2006). These variables were used to identify, on one hand, all the sites where fellings had not been done at all, and on the other, sites where fellings had not been done during at least the past 30 years. NFI plots on forest and other wooded land that have faced no fellings or other operations for at least 30 years or more (observations made on the field) were used to calculate a kernel density for ''naturalness'' using ArcGIS Spatial Analyst (McCoy and Johnston, 2001) expressed as the number of sites per km^2.

The silvicultural history (naturalness), expressed as density layer, was reclassified so that class 1 included sites where the density was below the overall mean for southern and central Finland, class 2 included sites where the density was between the overall mean and mean + 1 standard deviation (S.D.) and class 3 the sites where the density was higher than the overall mean + 1 S.D.

2.3.2. *Weighting*

When deriving the habitat index the input layers were assigned different weights to reflect their importance to biodiversity in the forests. Because these weights are not precisely known, however, we set equal weights for volume of growing stock, stand age and volume of CWD whereas productivity of the site and degree of naturalness received lower weights, the former because it was considered less important in the model and the latter because the layer was deemed somewhat imprecise. The final model for habitat index (HI) was as follows:

$$\mathrm{HI} = 0.25 \times \mathrm{VOL} + 0.25 \times \mathrm{AGE} + 0.125 \times \mathrm{BON} + 0.25 \times \mathrm{CWD} + 0.125 \times \mathrm{NAT}, \quad (1)$$

where HI: habitat index; VOL: estimated volume of growing stock; AGE: estimated stand age; BON: estimated productivity of the site; CWD: estimated volume of coarse woody debris above regional average; NAT: surrogate for the degree of naturalness.

Note that the maximum values of the variables are different (Table 1). Therefore, these weights mean that potentially we give the highest importance to the volume of CWD and volume of growing stock. The second highest importance is given to stand age. The next highest importance is given to potential productivity. The naturalness has the lowest importance in our model. Note also that we rescaled the habitat index value calculated according to Eq. (1) so that it ranges from 0 to 1 for easier interpretation of results. Thus, in Section 3 of this paper, the term "value of habitat index" indicates the rescaled value. After recalling, the average habitat index is 0.56 in the study region. The habitat quality model was build up to cover southern and central Finland and the data for the Satakunta area were afterwards extracted from the resulting habitat index layer.

2.4. *Cost-effective selection*

We next define a model that has a binding budget constraint for selecting areas into the conservation network (Ando et al., 1998; Balmford et al., 2000; Polasky et al., 2001; Juutinen et al., 2004). Because conservation funds are typically limited the idea of this cost-effective approach is to maximize biodiversity within a given budget. Thus, it is explicitly assumed that all the ecologically valuable targets are not protected, because it is too expensive. If a site is protected, it causes opportunity costs due to the lost harvesting revenues as loggings are prohibited in strictly protected areas. The available budget for conservation is taken as given, but by varying the size of the budget it is possible to reveal the trade-off between biodiversity benefits and costs of conservation. The larger is the budget, the larger are the conservation network and biodiversity benefits.

We formulate an integer linear object function, which includes the habitat index described in previous section (Eq. (1)). The objective function is presented in Eq. (2). It is maximized subject to constraints (3) and (4):

$$\max_{x_j} \sum_{i=j}^{m} \mathrm{HI}_j x_j \tag{2}$$

$$\sum_{j=1}^{m} c_j x_j \leq B \tag{3}$$

$$x_j = (0,1), \quad j = 1, \ldots, m \tag{4}$$

where HI_j is the value of habitat index of site j ($j = 1,\ldots, m$), $x_j = 1$ if site j is selected for protection and 0 otherwise, B the budget allowable for reserve network, c_j the opportunity costs of establishing reserve site j.

The object function (2) sums up the values of habitat index of the selected sites. Eq. (3) ensures that the sum of opportunity costs over the selected sites does not exceed the allowable budget. The constraint set (4) indicates that the choice variables must be binary. Thus, the sites are either protected or harvested.

Recall the value of habitat index is assessed for each site. However, it is likely that there is a certain threshold value for the habitat index defining which sites can be regarded as potential conservation targets. All forests do not need protection, but only those forest types that are rare and under a threat in the commercial timber production. The threshold can be incorporated into our model with the following equation:

$$\mathrm{HI}_j x_j \geq F, \quad j = 1, \ldots, m, \tag{5}$$

where F denotes the threshold value for the habitat index. Note that the habitat index consists of several components, i.e.

$$\mathrm{HI}_j x_j = \sum_{i=1}^{n} w_i \mathrm{HI}_{ij} x_j,$$

where HI_{ij} denotes the value of component i in site j and w_i denotes the weight of component i (see Eq. (1)). Accordingly, it is also possible to set more specific threshold values to the components of habitat index. Formally

$$\mathrm{HI}_{ij} x_j \geq F_i \quad i = 1, \ldots, n \text{ and } j = 1, \ldots, m, \tag{6}$$

where F_i denotes a threshold value given to the component i. By varying the thresholds values in Eqs. (5) and (6) we can construct alternative sets representing potential conservation targets. Then, using these different sets in the optimization and site selection we can compare the results associated with different thresholds values.

Recall the area of forestry land differs between raster cells. Therefore, we treat the value of habitat index as a unit value, i.e. value per hectare of forestry land, in the optimizations. Accordingly the total value of habitat index is 2.1 for a site having size of 3.5 ha forestry land and 0.6 unit value of habitat index, for example. We assume that the opportunity costs of protecting a given site (i.e. raster cell) depends on the timber production possibilities on this particular site. This practice is similar to the governmental land acquisition, where private landowners are compensated according to the commercial market values. More precisely, we multiply the volumes of different timber assortments (derived from the multi-source NFI thematic maps) by stumpage prices to calculate the value of standing timber for each site. The stumpage prices are presented in Table 2. The stumpage prices are same that the regional Forest Centre has used in 2003–2004 in the study area to value stands that private forest owners supplied for temporal conservation contracts (Gustafsson and Nummi, 2004). This practice does not take into account the expected growth of forest or land value, but typically these factors have only a marginal importance when the focus is on forests that are approaching to a mature age (age > 80 years). For these forests, the most important economic factor – affecting the benefit–cost ratio – is the value of standing timber.

To consider alternative sets for potential conservation targets we use specific thresholds values as described in

Table 2 – Stumpage prices

Timber assortment	€/ha
Pine sawtimber	47.10
Pine pulpwood	15.00
Spruce sawtimber	43.80
Spruce pulpwood	23.80
Birch sawtimber	37.90
Birch pulpwood	13.50
Other sawtimber	33.60
Other pulpwood	13.50

Table 3 – Threshold values for potential conservation targets

Criteria at site level	Set 1	Set 2	Set 3	Set 4
HI, value of the habitat index	>0.6863	>0.6863	>0.6863	>0.6863
AGE (years)	>40	>40	>40	>40
VOL (m^3/ha)	>50	>50	>50	>50
CWD (m^3/ha)	>2.5	>0	Not active	Not active
NAT (sites/km^2)	Not active	Not active	>0.072	Not active

AGE, stand age; VOL, volume of growing stock; CWD, volume of coarse woody debris above regional average; NAT, degree of naturalness.

Table 3. The threshold value for the habitat index itself ensures that the potential targets in each set have clearly higher ecological quality than the forest in average in the study area. More precisely, this threshold value is determined as mean plus one standard deviation. Similarly, the thresholds values for the volume of CWD and naturalness emphasize the importance of these factors in the site selection. The thresholds values for the stand age and stem volume of living trees (volume of growing stock) are used, because the forests with high biodiversity value are often mature or old-growth forests, in practice. However, we avoid setting too high threshold value for the stand age, as this would reduce the number of potential targets considerably. In addition, the threshold value for the volume of growing stock was set quite low level to ensure that the habitat types having low productivity are not excluded from the sets. Without using the threshold values for the stand age and volume of growing stock, many young forests or bare land sites would likely be selected into the conservation network, because the value of standing timber is very low at these sites. However, for these forest types it would be important to take also into account the expected forest growth when calculating the opportunity cost of a site, which was not done in this study. The information on the stand age and the volume of growing stock for each raster cell (site) were derived from the multi-source NFI thematic maps. The average volume of growing stock was 120 m^3/ha and the average stand age was 54 years in the study region.

The site selection model can be solved to find exact optimal solution by using a branch-and-bound algorithm, for example. However, the solving time may be long as the data set is large. Therefore, we used a heuristic procedure by simply ranking the sites according to benefit–cost ratios – value of habitat index/value of standing timber – and selecting the sites having the highest benefit–cost ratios into the conservation network under a given budget. The solutions of this simple procedure are optimal solutions under certain budget levels, but the all optimal solutions under all possible budget levels are not found using this procedure.

A classical approach in economics to describe trade-offs between productions of two alternative goods – with given inputs and technology – is to determine their PPF (Mas-Colell et al., 1995; see Fig. 3). It shows the maximum quantity of one good that can be produced given the quantity of the other good produced. Thus, the combinations of output on the PPF are technically efficient and the combinations under the PPF are inefficient. The combinations of output above the PPF needs more inputs than the economy has available. The farther away the PPF is from origin the larger is the production possibility set. The slope of the PPF reflects the opportunity costs of increasing the production of one good, i.e. how much the production of the other good is reduced. We applied this approach, but instead of using only quantities we measured the production of timber in monetary terms as there were several timber assortments those prices varied (Calkin et al., 2002; Boscolo and Vincent, 2003; Lichtenstein and Montgomery, 2003). The production of biodiversity was measured in terms of habitat index summed over the protected sells.

3. Results

3.1. Features of alternative sets of potential conservation targets

We first consider the basic features of determined sets of potential conservation targets. It is interesting to know how large these sets are and whether there are differences between sets in terms of habitat index and tree characteristics. These results are presented in Table 4.

A set of potential targets areas was found, covering about 13.7% of unprotected forestry land (Table 4, Set 4), focusing on forests that are approaching to a mature age and have a clearly higher habitat index than the forest on average within the study area. If we emphasize the volume of CWD along with the components used in Set 4, the area of potential targets reduces considerably depending on how high thresholds values are used for the volume of CWD (Table 4, Sets 1 and 2). The potential targets cover only 3% of unprotected forestry land, for a scenario where we require targets areas with a volume of CWD greater than 2.51 m^3/ha above regional average (along with the thresholds values used in Set 4). Setting thresholds values for naturalness (Set 3) reduced also the area of potential target compared with Set 4. In terms of habitat index, Sets 2–4 are rather similar, but Set 1 has a higher value than other sets. Similarly, Sets 2–4 have almost same mean stand age, but in Set 1 the mean stand age is somehow lower. Regarding, the mean volume of growing stock Sets 2 and 3 are quite similar. In the overall, the highest mean volume of growing stock is in Set 4 and the lowest in Set 1.

Table 4 – Key features of alternative sets of potential conservation targets

	Set 1	Set 2	Set 3	Set 4
Hectares	14,105	55,948	44,479	65,298
Mean HI	0.779	0.758	0.741	0.744
Mean VOL	160	173	174	179
Mean AGE	62	64	65	65

HI, habitat index; VOL, volume of growing stock; AGE, stand age.

Assessing statistical significances of the mentioned differences is a difficult task. The statistical significance of the different HI values, for example, can be roughly assessed using the ideas given in Section 2.2. HI is a function of volume and other quantitative variables. The relative error estimates can be assumed to be at least as high as that of volume, i.e. 4–5%. If we assume that the correlation coefficient of the estimates in the two sets is not more than 0.5, we can deduce that the standard deviation of the difference of the estimates is at least the standard error of the estimate (assumed same in here). This means that the single standard deviation of HI is about 0.1 and the differences are not statistically significant. This outcome does not mean, however, that using Sets 1–4 in site selection results in similar conservation networks and conservation costs.

3.2. Production possibility frontiers

We next analyse the trade-offs between biodiversity and timber production by using PPFs. PPFs for the different sets are depicted in Fig. 2. In addition to PPFs, Fig. 2 includes also the solutions representing the current management practice.

The estimated PPFs bring out two important features (Fig. 2.). The different thresholds values affect considerably the size of feasible production shown by the area under the PPF-curve. In this work, Set 4 cover the largest production possibilities set and Set 1 the smallest. Another important feature of PPF along with its position is its slope. It is often assumed that shape of PPF is concave indicating increasing marginal opportunity cost. In terms of our case study, the more areas are protected the more timber production one has to give up gaining an additional unit of habitat index. We can see, however, that the PPFs for Sets 1–4 are only weakly concave, and therefore, the marginal opportunity costs are

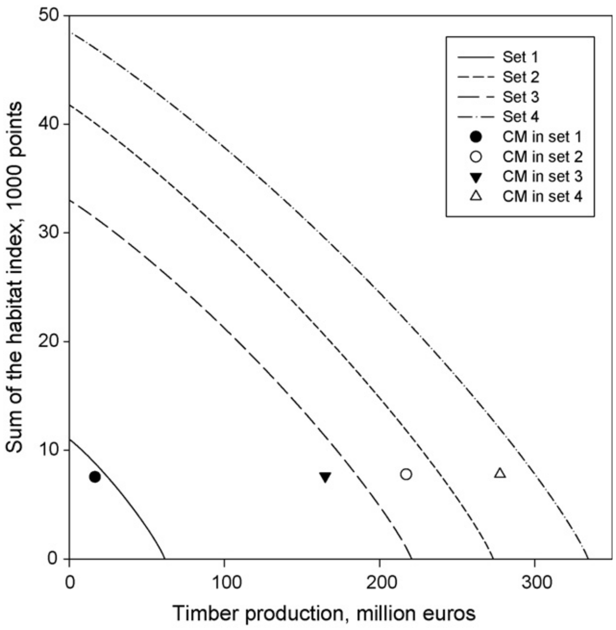

Fig. 2 – Production possibility frontiers for the different sets of potential conservation targets. CM denotes for current management.

increasing gradually. It is also interesting that the alternative PPFs seem to have quite similar slopes, although the size of feasible set differs remarkably between the sets.

These outcomes show that high thresholds values result likely in a case where the production possibility set is small. In practice, this simply means that if the conservation is restricted to the highest quality targets, it is not possible to achieve large conservation networks, as there are not enough high-quality targets. In particular, there is rather low number of sites having a high volume of decaying wood in the Satakunta region.

A socially optimal combination of the two outputs could be identified, if values to society were known for the both timber and biodiversity. This would be the point at which the slope of PPF is equal to the value to society of timber relative to biodiversity. Therefore, the weak concavity of the PFFs indicates that the optimal combination of the two outputs is sensitive to the relative values of these goods. In fact, it could be optimal to specialize and manage the forestry land exclusively for either biodiversity or timber (dominant use), if the differences between the values of these goods are large enough. However, the larger the production possibility set the more concave is the PPF indicating that it is probably optimal to manage the particular set for both biodiversity and timber (multiple use). The investigation of marginal opportunity costs illustrates also this outcome. For example, in the Sets 1 and 4 the marginal opportunity costs of increasing the sum of habitat index by 1 unit are about 1800 € when the first two sites are selected. When the last site is selected, the marginal opportunity costs are about 11,100 and 14,600 € in Sets 1 and 4, respectively.

The concavity of PPF reflects the differences in ecological and economic characteristics, i.e. benefit–cost ratios, between the sites, in our case study. If the characteristics of the sites were same, then the PPF would be a straight line, because we assumed that the sites are either protected or harvested (see Boscolo and Vincent, 2003). In other words, the joint production of the two goods was not possible at site level. Given the weak concavity of the PPFs, the variation of benefit–cost ratio between sites is rather narrow excluding the sites having the highest benefit–cost ratios.

The estimated PPFs can be used to evaluate the efficiency of current management practices. In Finland, forest reserves have typically selected by ranking the potential areas according to their ecological characteristics (along with criteria related to the presence of red-listed species and the location of areas, for instance; Alanen, 1992; Virolainen et al., 2001). Taking this approach into the context of our study, we can approximate the current management practice by assuming that protected areas cover 9285 ha including the areas that have the highest values for the habitat index in each particular production possibility set. Accordingly, the current management practice is clearly inefficient as the solutions representing the current management are under the PPFs (Fig. 2). The result suggests that the environmental managers could reduce conservation costs about 8–34% and still achieve the aimed target level of conservation or could establish about 11–38% larger conservation network under the given budget by taking both ecological and economic characteristics of the sites into account in the site selection instead of focusing only on ecological characteristics of sites.

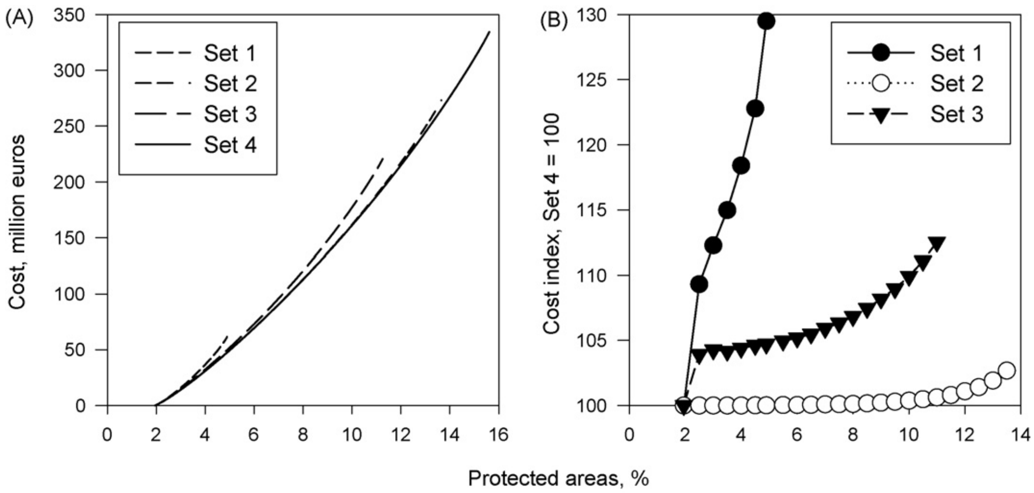

Fig. 3 – Absolute (panel A) and relative (panel B) costs of selected conservation network picked by using different sets for potential conservation targets.

3.3. *Cost comparison*

In this section, we compare the costs of conservation between different sets of potential targets. We plot the costs of different sets as a function of share of protected areas. The results are depicted in Fig. 3.

The cost differences between the estimated production possibility sets may be remarkable (Fig. 3B). For example, when about 4.5% of forestry land is protected, the costs of Set 1 are about 9 million € and 23% larger than in other sets. Similarly, when protected areas cover about 10% of forestry land, the cost difference between Sets 3 and 4 is about 16 million € and 10%.

Costs comparison reveals also several interesting features (Fig. 3A). First, the smaller is the considered production possibility set the higher are the costs caused by conservation. For instance, the costs curve of Set 1 situates above the cost curve of Set 4. This reflects the trade-off possibilities between ecological benefits and economic costs in a given set. The smaller is the set the smaller are these trade-off possibilities which increases costs of conservation, because it is not possible to found so many low-cost sites having high habitat index (see also Fig. 2).

Second, the costs differences between the sets increase as the conservation level increase. This indicates that typically the opportunity costs dominates habitat index in the site selection. In other words, the sites are more likely selected into conservation network due their low economic value than due their high ecological value. Recall that the value of habitat index may be relatively small at low-cost sites, because the volume of living trees affects positively to the value of habitat index. The dominance of the opportunity costs indicates that there is a larger variation in costs than in habitat index within the set of potential targets. For example, the relative standard deviation (R.S.D.) of the values of standing timber is 28.8% and the R.S.D. of the values of habitat index is 6.3% in Set 1. The figures in Set 4 are 27.3 and 5.3%, respectively.

Third, the thresholds values for alternative components of structural biodiversity affects differently to the costs. This suggests that some components of structural biodiversity can be protected without large cost-increments. For example, it will not cost much more to select sites where the volume of CWD is above the regional average instead of selecting areas without this requirement (see Set 2 and Set 4 in Fig. 3), but if we set tighter requirements for the volume of CDW (see Set 1 in Fig. 3) it will clearly increase the costs.

Overall, we can conclude that environmental managers could save in conservation costs by setting wider criteria for the potential conservation targets. These wider criteria involve a compromise in relation to biodiversity value and areas to be protected. However, a good decision would enlarge the size of production possibility set and increase the trade-off possibilities between ecological benefits and economic costs. The likely drawback of this approach is that the conserved targets may be of a lower ecological quality, however. This is particularly undesirable in the short term because of likely extinction dept in southern Finland (Hanski, 2000). In the long run, protecting sites of lower quality and low costs may turn out cost-efficient as forest succession effectively restores structures that are important for biodiversity (e.g. dead wood and large living trees) if given enough time.

3.4. *Quality of conservation networks*

We use habitat index to measure components of structural features of considered areas. This habitat index can be interpreted as describing the ecological quality of a given area. We next compare the average quality of selected conservation network between alternative sets for potential targets. For that purpose, we depict the average habitat index as a function of share of protected areas. Recall that the share of existing protected areas is about 2% in the study region.

The average habitat index increases as the share of protected areas increases (Fig. 4). This feature is same for all Sets 1–4. The increase is strongest for Set 1, which is determined according to the tightest thresholds values as the size of this set is the smallest. Moreover, the average habitat index of Set 1 increases rather gradually in contrast to

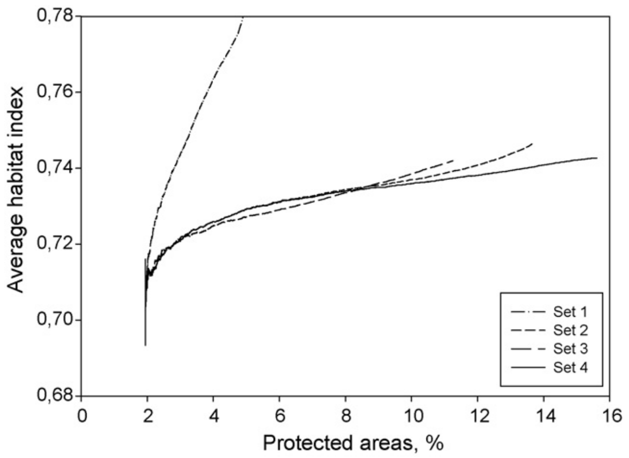

Fig. 4 – Ecological quality of selected conservation network picked by using different sets for potential conservation targets.

the other sets where the average habitat index increases first strongly but then the increase levels off.

These results show clearly that the sites selected at first have typically a relatively low value of habitat index. The average habitat index as a function of share of protected areas is a running average showing the trend in the change of the ecological quality of protected areas as the conservation network increases. Thus, typically, when this trend is increasing, the habitat index value of the next protected area is higher than the habitat index value of the previously protected area. The differences on average between the habitat index and the alternative production possibility sets are rather minor. Only the Set 1 have clearly higher average habitat index than the other sets. This means that relaxing the criteria for potential conservation targets does not lower much the ecological quality of the conservation network when the aim is to protect about 10% of forestry land in the study area, because the number of sites that fulfil tight ecological criteria is low. Results also show that 10% target is not feasible with ecologically high-quality sites (Set 1) because the number of sites that fulfil tight ecological criteria is simply too low in the current landscape. The 10% target can only be reached if requirements for ecological quality are less stringent.

4. Discussion

In this study, we investigated how alternative threshold values for the structural components of biodiversity affect cost-effective selection of protected areas in boreal forest. It was, therefore, important to consider both ecological and economic aspects of conservation. For this purpose, we revealed the trade-offs between the outputs of biodiversity and timber under alternative threshold values and examined them from several aspects.

We found that tight ecological criteria for potential conservation targets reduce the size of production possibility set considerably. Therefore, it may not be possible to achieve a large conservation network within the study area, as there are not enough high-quality targets fulfilling thigh criteria.

We also found that the PPF for the outputs of timber and biodiversity is only slightly concave when using the given threshold values. Boscolo and Vincent (2003) attained similar results by using a ‘‘proximity to climax’’ index for the biodiversity measurement. This index uses the structure of the old-growth forest as a reference point, and therefore, it basically represents a similar type of approach than the one used in this study. The weak concavity of PPF suggests that the optimal combination of the outputs is sensitive to the relative values of these goods. In the extreme, it could be optimal to manage the particular forest land exclusively for either biodiversity or timber, if the differences between the values of these goods are large enough.

One possible explanation for our findings may be that we did not take into account the interdependency between sites in biodiversity production. This is important because the selected conservation network should represent the whole spectrum of ecological features of a given area (Faith and Walker, 1996). For example, when maximizing the number of species in the selected conservation network, one should take into account that sites may cover same species, which typically results in a very concave PPF (Ando et al., 1998; Cabeza and Moilanen, 2001; Moilanen, 2007; Juutinen et al., 2008). Focusing on structural elements of biodiversity may result in a conservation network, which includes many similar sites (but see Juutinen et al., 2006). Consequently, the network potentially excludes part of ecologically important variation, e.g. in species and habitat types. This is because structural elements of biodiversity are typically measured using a scoring framework where the scores of sites, i.e. ecological value, do not depend on the scores of other sites (Pressey and Nicholls, 1989). Nevertheless, our study certainly helps to reconcile forest landscape conservation and economic value.

Our results suggest that an integrated approach in forest conservation could provide to environmental managers considerable costs savings compared with current management practices while promoting to conserve more land on a sustainable way. The results are in line with previous studies on integrated management in boreal forests (Juutinen et al., 2004; Hurme et al., 2007; Mikusinki et al., 2007). Setting appropriate criteria for potential conservation targets involves, however, a trade-off between cost savings and ecological quality of conserved areas. Relaxing the criteria result in large production possibility set and reduces the cost of conservation, but lower the quality of conserved areas. It seems, however, that the reduction in the quality may be minor, when there are only a small number of high-quality sites. Thus, it may be reasonable to use ‘‘loose criteria’’ for potential conservation targets and not protect the ecologically most valuable sites, when this causes large lost in harvest revenue. In this way larger areas can be protected with the given budget, which may be a viable option particularly if the target is ambitious (e.g. 10% of forest land) in relation to present level of conservation (Mönkkönen et al., in press). In addition, it could be more efficient to restore the ecological values of the low-cost sites than protect the high costs sites. For example, Ranius et al. (2005) showed that it is very expensive to increase CWD in managed forest by prolonging the rotation period (temporal protection) compared with other management measures, such as artificial creation of high stumps.

Our study hopefully offers insights for future work. For example, in this study we did not consider the location of selected areas in the analysis. For this purpose, it would be important to identify which landscape elements are the most critical for the maintenance of overall forest landscape continuity and connectivity (Pascual and Saura, 2006). Spatial configuration of protected stands may be an important issue in fragmented landscapes where individual dispersal among habitat patches is limited, and a rule-of thumb recommendation is to spatially aggregate selected areas whenever possible (Wilson and Willis, 1975). However, in boreal forest landscapes, where forest succession continuously alters stand and landscape characteristics, there is not much evidence that fragmentation affects species persistence (e.g. Schmiegelow and Mönkkönen, 2002). Therefore, habitat availability, not the spatial configuration, is the primary concern (Andrén, 1994; Fahrig, 1998). It is possible to extend our approach to cover also the spatial configuration of protected areas, but one needs more sophisticated methods to solve explicitly spatial site selection problems (e.g. Siitonen et al., 2002). Moreover establishing compact conservation networks in Southern Finland may be difficult as the forests are mainly privately owned and the forest ownership is fragmented with small stand size. Regarding our results, it is likely that the restrictions or targets on the configuration of conservation network have a stronger influence when tight criteria are used for potential conservation targets than when loose criteria are used. Thus, including the configuration into our analysis could increase the cost differences between the production possibility sets. The larger the set, the easier it is to meet the given targets on configuration. In this sense, large regional planning will be advisable to improve the conservation network and connectivity of valuable sites.

Another important consideration is the inherent complexity of developing a habitat index. There is no a single way or recipe, most of the literature on habitat quality index provides specific information for a particular condition or a particular need or application. What it becomes difficult then is to reach a generalisation level of an index that can describe the biodiversity value of the landscape. Lacking detailed ecological knowledge, the choice of variables and their weights for constructing a habitat quality index involves always some subjective judgements. The variables used in this study reflect, however, directly and indirectly many features that are regarded important in conserving forests in Southern Finland (Ympäristöministeriö, 2003). However, conservation decisions are not taken on the basis of a single habitat index, but considering several others criteria, such as endangered species for different forest areas, individuals attitudes towards particular species and their stated willingness to allocate funds for their conservation among others (Martin-Lopez et al., 2007; Maiorano et al., 2007; Rondinini and Pressey, 2007). Moreover, comparing different production possibility sets, which represent different threshold values of the ecological criteria, can also be interpreted as a sensitivity analysis of the weighted values. Finally, constructing a habitat index using remote sensing data involves certain limitations. In particular, the dead wood predictions used in this study are large area averages indicating that the deviations from the local dead wood volumes may be high.

It must be considered as well that most landscapes provide a multitude of functions and are subject to many possible land uses (Groot, 2006). Usually different combinations of land uses are possible, so thresholds values can be adjusted in combination with the prioritization of multi-use of forest, respectively. In this sense, to analyse the various planning and management alternatives for multi-functional landscapes, in particular for multi-uses of forest, may be a valid alternative in order to improve the habitat index.

In all, given the widespread importance of forest quality habitat loss and forest degradation as a threat to biodiversity, our results highlight the value of combined economic needs and biodiversity value. We believe that the integration of economic and ecological values and a strong compromise to protect valuable habitat are essential to avoid biodiversity loss towards the 2010 target. We conclude that extending similar methods to many other areas would be a very valuable contribution to a sustainable forest management.

Acknowledgments

This article is part of the Biodiversity and Monitoring Program in Finland (MOSSE). Funding from the Ministry of the Environment is gratefully acknowledged.

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