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Cost-effectiveness of conservation strategies implemented in boreal forests: The area selection process

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ARTICLE INFO

Article history:

Received 29 April 2008

Received in revised form

19 November 2008

Accepted 23 November 2008

Available online 6 January 2009

Keywords:

Biodiversity

Species–investment curves

Information cost

Opportunity cost

Complementarity

ABSTRACT

To protect land from commercial exploitation is a common conservation practice. However, this requires large financial resources and it is therefore important to evaluate the cost-effectiveness of different strategies used in the selection of these conservation areas. In this study we compare four strategies and relate the differences in cost-effectiveness to differences in the selection process. We measure conservation benefits both as the amount of three tree structures and as the number of species in three species groups. We also estimate both the information cost associated with selecting conservation areas and the opportunity cost. We found the key habitat strategy to be the over-all most cost-effective. In this strategy, the areas have a flexible size and are selected by the authorities in a national field survey. The least cost-effective strategy was one where the selection was based only on forest classes in a satellite map. Intermediate were the retention group strategy, where small areas are left by the forest owner at harvesting, and the nature reserve strategy, where large areas are selected by the authorities. We emphasize that the differences we found are associated with the selection process and that other aspects, such as long-term survival of species, may rank the strategies differently. We conclude that the cost-effectiveness of a selection strategy depends on the size of the planning area for selection of conservation areas, the size of the conservation areas, the objective of the agent making the selection, and the amount and type of information on which the selection is based.

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1. Introduction

Unsustainable use of natural resources poses a threat to biodiversity worldwide (e.g. Sala et al., 2000). As biodiversity has values that are currently not reflected on regular markets, policy tools are required to regulate activities with potential negative impacts on biodiversity (Convention on Biological Diversity, 1992). One of the most common conservation measures (at least in natural ecosystems) is to pro-

tect land from commercial exploitation. From the perspective of the policy maker, different strategies can be used to create such conservation areas and these may differ in several ways, for example in the use of voluntary or mandatory policy tools (Merlo and Paveri, 1997), in how conservation areas are selected, in the degree of protection, or in management directives. As protection of land requires large financial resources, both public and private, it is important to evaluate the cost-effectiveness of different

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doi:10.1016/j.biocon.2008.11.014

strategies (Saterson et al., 2004; Wätzold and Schwerdtner, 2005; Naidoo et al., 2006; Murdoch et al., 2007).

In analyses of cost-effectiveness, the costs of conservation are expressed in monetary terms whereas the benefits remain in the original units (e.g. numbers of species). The most cost-effective strategy is the one that reaches a given conservation level for the least cost or, alternatively, maximizes the conservation level for a given cost (Ando et al., 1998). Conservation levels can be defined in terms of several different parameters (e.g. protected area, population size or number of species), and conservation costs can be of several different types (e.g. opportunity costs, transaction costs, and management costs) (Naidoo et al., 2006). Given this, it is important that analyses of conservation strategies include all major costs (McCann et al., 2005) as well as several different aspects of biodiversity.

The process of selecting conservation areas has recently received much attention within the framework of reserve selection and conservation planning (e.g. Margules and Pressey, 2000; Cabeza and Moilanen, 2001). However, these studies have so far focused on hypothetical strategies and, although important, such studies necessarily entail large uncertainties regarding both the actual outcome for biodiversity and the economic consequences. Studies evaluating the cost-effectiveness of actually implemented strategies are on the other hand rare (Millennium Ecosystem Assessment, 2005) and the importance and great value of such studies is now being acknowledged (Sutherland et al., 2004; Fazey et al., 2005; Ferraro and Pattanayak, 2006).

To understand the reasons for differences in cost-effectiveness between strategies we must consider the implications that the strategies have for the selection process. One important factor here is the size of the planning area from which conservation areas can be selected. If a region consists of one large planning area the best conservation areas of that region can be selected. However, if the region is divided into many small planning areas we can only select the best conservation areas within each planning area. Larger planning areas therefore lead to more cost-effective selections (Strange et al., 2006). The required size of conservation areas is also important as small conservation areas or conservation areas with flexible size can be located in a more cost-effective way within a planning area (Groeneveld, 2005; Ranius and Kindvall, 2006). Furthermore, the interest of the agent responsible for the selection of conservation areas is important. Whereas the interest of conservation agencies is to maximize conservation value, forest owners can be expected to also consider other benefits such as economic profit and personal values.

Another important aspect is the amount of information on which the selection is based. This factor is, in fact, linked to the previous ones, as the size of the planning area, the size of the conservation areas and the interest of the selecting agent influence the amount and type of information that is collected. If the planning area is small or conservation areas large there will be fewer possible conservation areas to choose from and less information will be required compared to when planning areas are large or conservation areas small. Furthermore, agents with less interest in conservation will collect less information about conservation value compared to other interests. In connection with this we notice that the cost of obtaining information is the most relevant transaction cost for the selection process (Kolstad, 2000). At the same time, lack of informa-

tion also leads to costs in terms of sub-optimal and less cost-effective selections of conservation areas (Freitag and Van Jaarsveld, 1998; Richardson et al., 2006; Grand et al., 2007).

In this study we evaluate the cost-effectiveness of three conservation strategies implemented in Swedish forests: the nature reserve strategy, the woodland key habitat strategy, and the retention patch strategy. We focus on how the differences between the strategies influence the selection process and how this in turn affects cost-effectiveness.

For nature reserves and key habitats the planning areas are large, at least entire counties, whereas retention patches are selected within a single forest stand. Furthermore, nature reserves are large (usually >20 ha), key habitats have a medium size (average 3 ha), and retention patches are small (usually <0.5 ha). Nature reserves and key habitats are identified by authorities with conservation interests whereas forest owners are responsible for the identification of retention patches (see Box 1 for more details about the strategies).

Box 1. Three conservation strategies implemented in Swedish forests.

Nature reserves are established by the County Administrative Boards (Länsstyrelserna) and are generally larger areas (>20 ha). They consist mainly of areas with high conservation value but can also, in order to achieve a coherent area, contain areas with lower value. They are identified in a complex process based on information from the Swedish Forest Agency, forest companies, private persons, and non-governmental organizations (NGOs), followed by a brief inventory of forest structures and, occasionally, species. The nature reserves are either bought by the state or the land owner is compensated for future loss in income.

Woodland key habitats are identified in a large national survey initiated by the Swedish Forest Agency. These are areas with high conservation value, often with old-growth characteristics and potential to host red-listed species (Nitare and Norén, 1992). They are identified based on stand characteristics, occurrence of red-listed and indicator species, and stand history (Norén et al., 2002). Key habitats can essentially have any size but the average is 3 ha (Swedish Forest Agency, 2007a). They are not all formally protected but should be the first areas to set aside either voluntarily by the forest owners, as a part of their certification commitments, or legally, in which case the owner is compensated by the state.

Retention patches are set aside by the forest owners according to the Swedish Forestry Act (30 §), stating that habitats important for red-listed species and biodiversity in general should be left intact at final felling (e.g. bogs, ravines, old trees, dead wood, and areas around streams, wells, rocks, and cliffs) (SKSFS, 1993). However, the law only requires a level of retention that does not significantly reduce stand value (in practice, not more than 5% of the stand volume has to be retained). Furthermore, voluntary certification standards require retention of 10 trees per ha, including the ones retained according to the law (Anon., 2000). The identification and delimitation of retention patches is based almost exclusively on structures and is done by the forest planner when planning for final felling. Apart from the retention requirement these areas have no formal protection.

The overall Swedish model for conservation of forest biodiversity is built on the assumption that the different strategies complement each other and so reach a cost-effective and functional whole (Swedish Environmental Protection Agency, 2005). This is based on the need for multi-scaled conservation measures, as both biodiversity, ecological processes, and management occur on multiple scales (Lindenmayer et al., 2006). Thus, the ranking of the strategies would be expected to differ between, for example, different species groups. Furthermore, cost-effectiveness is expected to increase when strategies are combined.

The first aim of our study is to estimate the cost-effectiveness of the strategies with respect to the selection of conservation areas. We compare the three implemented strategies both to each other and to a hypothetical strategy where conservation areas are chosen based on information from a satellite map. The first questions we address are consequently: (1) How cost-effective are the three implemented strategies, relative to each other, and to the hypothetical strategy? (2) How are differences in cost-effectiveness related to differences in the selection process? (3) How much does the information cost influence the cost-effectiveness of a strategy? The second aim is to evaluate to what degree the strategies complement each other and the two last questions we ask are therefore: (4) Does the ranking of the strategies differ for different aspects of biodiversity? (5) Can cost-effectiveness be increased by combining strategies?

2. Materials and methods

2.1. Study area and forest type

To answer the questions we carried out a field study in the county of Gävleborg in middle boreal Sweden. The study area measured approximately 150×150 km with central position $61^{\circ}45'N$, $16^{\circ}10'E$. In Gävleborg 80% of the land area is covered with forest, 45% of the forest is privately owned, the average standing tree volume is 138 m^3 per ha and 49% of the volume is made up of Scots pine (*Pinus sylvestris*), 35% of Norway spruce (*Picea abies*), and 10% of birch (*Betula pendula* and *B. pubescens*). The average cutting age is 105 years (data compiled by the Swedish National Forest Inventory) and there is on average 9 m^3 dead wood per ha (Swedish Forest Agency, 2007a).

We studied old spruce forests on mesic to moist soil. This is a forest type which is very valuable, both to the forest industry and from a conservation perspective. Finding cost-effective conservation strategies for this type of forest is therefore of great importance. To identify areas of old spruce forest we used the satellite map “wRESEx”, which shows different forest classes in 25×25 m resolution (Angelstam et al., 2003). In the study area we found 160 000 ha of non-swamp forests with >70% spruce, and out of this area 12% was more than 110 years old. The old spruce forest was very fragmented; the largest coherent area was 30 ha but the average size was only 0.2 ha. By combining wRESEx with GIS-layers showing the conservation areas (provided by the county administrative board of Gävleborg and the Swedish Forest Agency) we found that 2% of the old spruce forest was classified as key habitat (data only for privately owned forests) and

4% was within nature reserves. We were not able to find out how much old spruce forest was set aside as retention patches as these often are too small to be identified in wRESEx.

2.2. Study sites

To identify the proper forest type we used the following criteria in wRESEx: >70% spruce, >110 years, >5 km from the coast, <500 m above sea level, and on mesic to moist soil. Potential sites in nature reserves and key habitats had to be larger than 50×50 m and were identified using the GIS-layers mentioned above. Retention patches were defined as groups of trees smaller than 0.5 ha and completely surrounded by a clear-cut area created less than 10 years ago. Potential sites in such areas had to have a minimum size of 25×25 m and were identified using information from the Swedish Forest Agency and the forest company Stora Enso. From the potential sites we randomly selected 20 study sites of each type of conservation area. We also made a random selection of 20 sites larger than 50×50 m of the same forest type but neither included in reserves nor classified as key habitat. These sites represent the hypothetical strategy, where areas are selected based only on information from a satellite map, and are here referred to as “satellite-identified areas”. Reserves and key habitats were not included as the data set was originally collected for a slightly different purpose (see Perhans et al. (2007) and Djupström et al. (2008), where the satellite-identified areas are referred to as “old managed forest”), but since such areas make up only 6% of the old spruce forest in the study area the difference should be small. For all strategies the requirement was that study sites should be >1 km apart.

2.3. Field data

After the 80 study sites had been selected, we placed a circular study plot with 10 m radius in the centre of each site. These plots were then inventoried with respect to forest variables, biodiversity-related structures and species richness in June–October 2004. For each plot we recorded site quality index (Hägglund, 1979), vegetation type (Hägglund and Lundmark, 1981), thinning history, number of living trees with >10 cm diameter at breast height (dbh), dbh of those trees, and height and age of the two tallest trees in the plots. The standing volumes of living trees were then estimated from growth functions by Ekö (1985). We also measured all standing and lying dead trees and tree parts with ≥ 10 cm diameter as well as the amount of bark on them and their stage of decay (Djupström et al., 2008). From the collected data we estimated the volumes of three biodiversity-related tree structures known to be rare in mature managed forests: dead wood, deciduous trees and large-diameter trees (Berg et al., 1994).

The presence of bryophyte species on all substrates was recorded, as well as the presence of lichen species on living and dead spruce trees, up to a height of 2 m (Perhans et al., 2007). We also sampled saproxylic beetles (i.e. dependent on dead wood). However, these have a very clumped distribution and it was therefore not possible to make an inventory of the whole plots. Instead beetles were sampled by sieving 0.5 m^2

bark from each of five dead lying or standing spruce trees in early stages of decay (for more details see [Djupström et al. \(2008\)](#)). Data from the five bark samples in each of the plots were pooled. For each species group we analysed both the total number of species and the number of species red-listed in Sweden ([Gärdenfors, 2005](#)).

2.4. Opportunity cost

We estimated the cost of protecting each study plot as the opportunity cost, which at the stand level equals foregone timber profits in terms of net present value. To calculate this value we used our collected forest variables and the computer program Plan 33 ([Ekvall, 2001](#)). All simulations of stand management were made in order to maximize the net present value ([Faustmann, 1849](#)) under the assumption of 3% real interest rate. Price lists in Plan 33, for timber and pulpwood as well as costs for silvicultural measures, were obtained from Mellanskog, a forest owner co-operative (cf. [Ekvall, 2001](#); [Ranius et al., 2005](#)).

2.5. Information cost

The information cost is the cost for obtaining the information on which the selection of conservation areas is based. To get the information cost for retention patches, we asked two private forest companies (Holmen and Korsnäs), the state-owned forest company Sveaskog and the Swedish Forest Agency to estimate this cost. For key habitats we used the sum spent by the Swedish Forest Agency for identification of key habitats in Gävleborg county, and divided this with the area identified as key habitat. For nature reserves, the County Administrative Board in Gävleborg estimated the information cost. However, due to the complex identification process for reserves this estimate is uncertain. The selection of conservation areas from the satellite map was assumed to have negligible information cost as satellite maps like wRESEx already exist for the whole country and the selection of areas is done without any further information.

2.6. Data analysis

2.6.1. Background data

We first compared the strategies with respect to twelve parameters recorded on a per plot basis (living tree volume, area of bark on spruce in early stage of decay, volume of the three biodiversity-related tree structures, number of species in the three species groups, number of red-listed species in the three species groups, and opportunity cost) using ANOVA followed by Tukey HSD in the program Statistica ([StatSoft, 2005](#)). When data did not comply with the assumptions of ANOVA we complemented with Kruskal–Wallis test.

2.6.2. Cost-effectiveness

We then compared the strategies with respect to how cost-effective they were for each of the nine biodiversity parameters (volume of three tree structures, number of species in three species groups, and number of red-listed species in three species groups). For tree structures we assessed cost-effectiveness by means of the ratio between the amount of

a structure and the opportunity cost of a plot, and compared the strategies with the same analysis as above.

However, number of species does not increase linearly with opportunity cost but with a diminishing rate. The comparison of the cost-effectiveness of the different strategies must therefore be done at the same investment level or, alternatively, at the same number of species. We used sample-based species accumulation curves to calculate the average number of species for each number of sampled plots for each strategy ([Colwell, 2006](#)). We then multiplied the number of plots with the average opportunity cost for plots of that strategy to get species–investment curves for bryophytes and lichens ([Naidoo and Adamowicz, 2005](#)). This way we could estimate how many species would, on average, be included at different opportunity costs.

Because the number of beetle species was not recorded per plot but per 2.5 m² of spruce bark we divided the average opportunity cost per plot with the average amount of such bark per plot and multiplied with 2.5 in order to determine the opportunity cost of a forest area containing 2.5 m² spruce bark for each strategy. These values were then multiplied with the number of plots in the species accumulation curves to get a species–investment curve for beetles.

To compare the strategies we used the estimates of standard error associated with the sample-based species accumulation curves ([Colwell et al., 2004](#)). We calculated the confidence intervals for the curves and then considered strategies to be significantly different when the confidence intervals did not overlap. As 95% confidence intervals have been found to be too conservative ([Schenker and Gentleman, 2001](#)) we used 84% confidence intervals. These yield a probability of overlap of 0.95 when standard errors are approximately equal ([Payton et al., 2003](#)). However, these significance tests are rough approximations as the estimated standard errors do not take the variance in opportunity costs between plots into account. Still, as no method accounting for this is yet available (Robert K. Colwell and Chang Xuan Mao, pers. com.) we regard this as the best possible option.

We summarized the results for the biodiversity parameters by ranking the four strategies with respect to cost-effectiveness for each of the nine parameters and then calculated the average of the nine ranks, the standard error, and the 84% confidence interval. To see if the differences in information costs influenced the results we repeated the whole cost-effectiveness analysis with information costs added to the opportunity costs.

2.6.3. Combination of strategies

We examined whether differences in species composition (complementarity) between the different strategies could make a combination of two strategies more cost-effective than the better of the two strategies alone. This was done at two levels. We first looked whether different strategies would be most cost-effective for different biodiversity parameters by comparing the cost-effectiveness values and the species–investment curves. We then looked for differences within species groups by analysing all possible combinations of two strategies, i.e. six different combinations. For each combination we randomly selected two plots from each of the two strategies and recorded the number of species in the different

species groups. This was repeated 2000 times and the average number of species for the combination was recorded. The same was then done with a random selection of 4, 6, 8, and 10 plots from each of the two strategies in each combination. From this we constructed species–investment curves which showed the average number of species at different investment levels when equally many plots were selected from each of the two strategies in a combination. These curves were compared to the species–investment curves of the pure strategies and a new ranking, which included all ten strategies (four pure and six combined), was then done for each of the six species groups.

3. Results

3.1. Background data

Both the total volume of living trees per ha and the opportunity cost was lower in retention patches than in the other types (Table 1). The volume of deciduous trees was higher in nature reserves than in satellite-identified areas, whereas

the volume of trees larger than 30 cm dbh was lower in retention patches than in the other types. There were no differences in the volume of dead wood.

For both beetles and bryophytes, the number of species per plot was higher in key habitats than in satellite-identified areas and retention patches (Table 1). For lichens there were no differences. The number of red-listed species was higher in key habitats than in retention patches for all three species groups (Table 1).

3.2. Cost-effectiveness

For large-diameter trees, retention patches were significantly less cost-effective than the other conservation areas (Table 2). For deciduous trees, nature reserves were more cost-effective than satellite-identified areas, the other strategies being intermediate. For dead wood, there was no significant difference in cost-effectiveness between the four types.

The species–investment curves show the cost-effectiveness of the different strategies with respect to the different species groups (Fig. 1). Retention patches and key habitats

Table 1 – Differences in parameters associated with costs and biodiversity parameters of conservation strategies.

	Nature reserves	Key habitats	Retention patches	Satellite-identified areas	p-value
<i>Cost-related parameters</i>					
Tree volume ($\text{m}^3 \text{ha}^{-1}$) ^A	348 ± 23 a	323 ± 28 a	201 ± 20 b	313 ± 20 a	<0.001
Bark area ($\text{m}^2 \text{ha}^{-1}$)	253 ± 47	373 ± 63	513 ± 160	185 ± 38	0.065
Opportunity costs (€ ha^{-1})	5378 ± 474 a	5277 ± 591 a	3048 ± 415 b	5753 ± 515 a	0.001
Bark opportunity cost (€ 2.5 m^{-2})	53	35	15	78	
Information cost (€ ha^{-1})	56	121	32	0	
<i>Biodiversity-related structures</i>					
Trees >30 cm dbh ($\text{m}^3 \text{ha}^{-1}$)	119 ± 21 a	157 ± 30 a	33 ± 13 b	125 ± 24 a	0.002
Deciduous trees ($\text{m}^3 \text{ha}^{-1}$)	58.0 ± 10.9 a	36.6 ± 11.6 ab	25.6 ± 7.4 ab	13.2 ± 4.3 b	0.007
Dead wood ($\text{m}^3 \text{ha}^{-1}$) ^A	31.8 ± 6.1	28.2 ± 3.9	22.9 ± 6.4	22.5 ± 5.7	0.594
<i>Species per plot^A</i>					
Beetles	17.9 ± 1.0 ab	19.2 ± 1.2 a	15.0 ± 1.2 b	15.2 ± 0.9 b	0.015
Bryophytes	47.0 ± 3.0 ab	59.6 ± 4.4 a	40.6 ± 3.2 b	38.1 ± 2.2 b	<0.001
Lichens	58.6 ± 1.2	62.4 ± 1.7	57.6 ± 1.7	59.0 ± 1.5	0.140
<i>Red-listed species per plot^A</i>					
Beetles	0.30 ± 0.11 ab	0.75 ± 0.19 a	0.10 ± 0.07 b	0.75 ± 0.23 a	0.011
Bryophytes	0.90 ± 0.23 a	0.95 ± 0.22 a	0.10 ± 0.07 b	0.15 ± 0.11 b	<0.001
Lichens	1.25 ± 0.23 ab	1.40 ± 0.26 a	0.60 ± 0.13 b	0.80 ± 0.12 ab	0.014

Mean ± SE of 20 plots from each strategy. The p-values presented are from ANOVA but Kruskal–Wallis tests gave similar results. Significant p-values (≤ 0.05) are in bold. Means followed by the same letter were not significantly different ($p > 0.05$) according to Tukey HSD tests.

A These results are also presented in [Perhans et al. \(2007\)](#) and/or [Djupström et al. \(2008\)](#).

Table 2 – Differences in cost-effectiveness for three biodiversity-related tree structures, measured as the ratio between the amount of a structure and the opportunity cost.

	Nature reserves	Key habitats	Retention patches	Satellite-identified areas	p-value
Trees >30 cm dbh ($\text{m}^3 1000 \text{€}^{-1}$)	23.5 ± 5.1 a	27.2 ± 4.0 a	7.1 ± 2.5 b	17.9 ± 2.7 a	0.002
Deciduous trees ($\text{m}^3 1000 \text{€}^{-1}$)	13.4 ± 2.6 a	8.7 ± 3.0 ab	11.2 ± 3.5 ab	2.6 ± 0.7 b	0.034
Dead wood ($\text{m}^3 1000 \text{€}^{-1}$)	6.5 ± 1.2	6.7 ± 1.4	10.5 ± 3.1	4.3 ± 1.2	0.134

Mean ± SE of 20 plots from each strategy. The p-values presented are from ANOVA but Kruskal–Wallis tests gave similar results. Significant p-values (≤ 0.05) are in bold. Means followed by the same letter were not significantly different ($p > 0.05$) according to Tukey HSD tests.

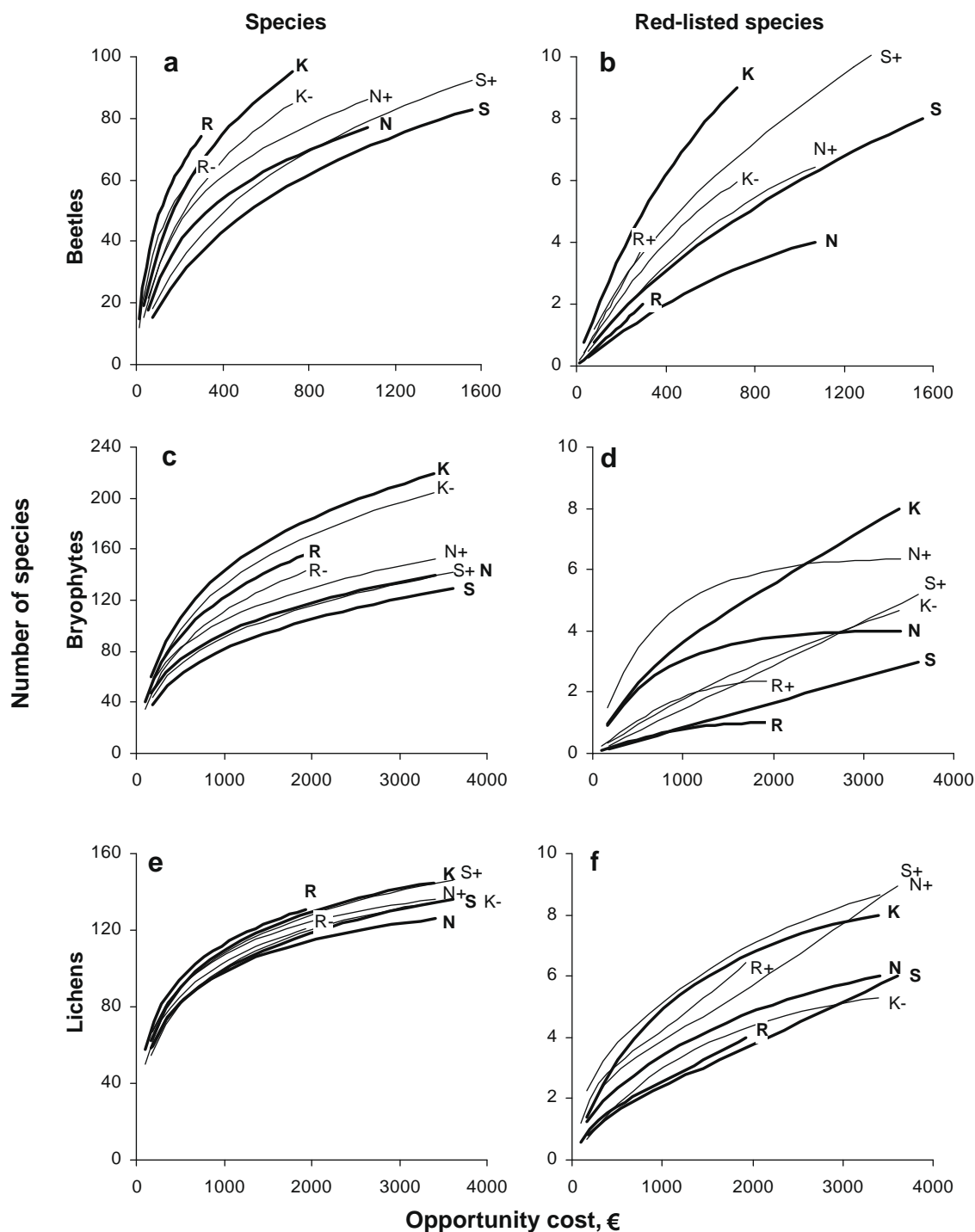


Fig. 1 – Cost-effectiveness in terms of average number of species included in relation to cost (species–investment curves) in nature reserves (N), key habitats (K), retention patches (R), and satellite-identified areas (S) (thick lines), and 84% confidence intervals relevant for determining statistical significance (thin lines; + for upper bound, – for lower bound). The data are from 20 study plots in each strategy.

were the most cost-effective strategies (had the highest number of species at any given cost or the lowest cost at any given number of species) for all species of all three species groups (Fig. 1a–c), but for lichens the differences were small and not significant. For red-listed species, key habitats were significantly more cost-effective than nature reserves for beetles but no other differences were significant (Fig. 1d–f).

When summarizing the results for all biodiversity parameters, we found key habitats to be the most cost-effective strategy (Fig. 2). This strategy had the highest average cost-effectiveness rank and belonged to the group of most cost-effective strategies for all of the five parameters where there were significant differences between the strategies. The retention patch strategy had the second highest average rank

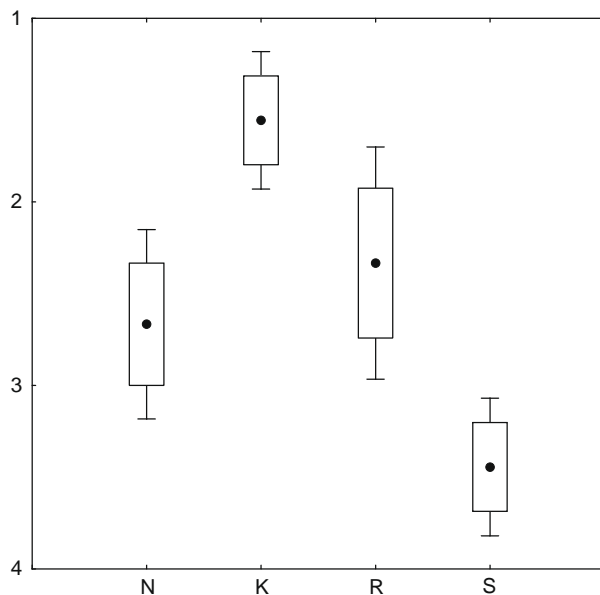


Fig. 2 – Over-all cost-effectiveness of nature reserves (N), key habitats (K), retention patches (R), and satellite-identified areas (S), expressed as average rank over nine biodiversity parameters; mean (dot), \pm SE (box), and \pm 84% confidence interval (whiskers). The strategy with the lowest value is the most cost-effective.

and was among the most cost-effective strategies for four of the five parameters. Nature reserves and satellite-identified areas were both among the most cost-effective strategies for two of five parameters, but nature reserves had a higher average rank.

The information cost for retention patches was estimated to between 24 and 42 € per ha retained forest, with an average of 32 € per ha (Table 1). For the identification of key habitats the Swedish Forest Agency spent 745 000 € in the Gävleborg county between 1993 and 2005 (Per-Olof Ståhl, pers. com.). This resulted in 6175 ha of forest being recognized as key habitats, i.e. an information cost of 121 € per ha. The information cost for nature reserves was estimated to 56 € per ha. Thus, the information costs were very low in comparison to the opportunity costs (i.e. between 0% and 2%) and did not influence the ranking of the strategies.

The cost-effectiveness of combinations of two strategies were generally intermediate between the two, except for

red-listed bryophytes where a combination of 10 nature reserves and 10 satellite-identified areas was slightly more cost-effective than 20 nature reserves, and for red-listed lichens where a combination of retention patches and satellite-identified areas was less cost-effective than either pure strategy (Table 3).

4. Discussion

The key habitat strategy was the most cost-effective strategy in our study, followed by retention patches and nature reserves, and all three implemented strategies were generally more cost-effective than the hypothetical selection of conservation areas from a satellite map. Thus, our results suggest that it is wise to continue with the identification and protection of key habitats, especially as it is estimated that only about 20% of the key habitat area in Sweden has been identified so far (Swedish Forest Agency, 2008) and out of that area only about 27% has been formally protected (Swedish Forest Agency, 2007b).

The only objective in the key habitat strategy is to select areas with high conservation value and to meet this objective the Swedish Forest Agency carried out a specific inventory. As a result, our study showed that key habitats had the highest number of species per plot for all species groups and the largest amount of large-diameter trees. On the other hand, opportunity costs are not considered in the selection of key habitats and, accordingly, we found that opportunity costs did not differ from the random selection of old spruce forests from the satellite map.

The objective in the nature reserve strategy is, just as for key habitats, to select areas with high conservation value, but in addition the areas should be large. As the old-growth forest is very fragmented in Sweden, it can be difficult to combine these two objectives and therefore areas with lower conservation value are often included (Ranius and Kindvall, 2006). In our study, reserves had the highest amount of deciduous trees and dead wood, whereas the amount of large-diameter trees and the number of species per plot was intermediate. Although the differences in number of species between key habitats and nature reserves were not very pronounced at the level of plots, the over-all cost-effectiveness of the reserve strategy was considerably lower. The difference is mainly due to the higher complementarity among key habitats (the plots were more different in terms of which species they contained), resulting in a steeper species accu-

Table 3 – Ranking of cost-effectiveness of pure and combined strategies for each of the six species groups and the average of the six ranks. The strategy with the lowest value is the most cost-effective.

	N	K	R	S	NK	NR	NS	KR	KS	RS
Beetles	8	3	1	10	5	4	9	2	6	7
Bryophytes	8	1	3	10	4	6	9	2	5	7
Lichens	10	3	1	8	6	5	9	2	4	7
Red-listed beetles	10	1	8	5	4	9	7	2	3	6
Red-listed bryophytes	6	1	10	8	2	7	5	4	3	9
Red-listed lichens	5	1	8	9	2	6	7	3	4	10
Average rank	8.1	1.6	5.6	7.9	3.9	6.6	7.6	2.4	4.0	7.4

N = nature reserves, K = key habitats, R = retention patches, and S = satellite-selected areas.

mulation curve and a higher total number of species than for reserves (Perhans et al., 2007). This might be because key habitats are focused on small distinct habitats such as forests close to streams, whereas nature reserves could include old-growth forest in general. As for key habitats, opportunity costs are not considered in the selection of nature reserves, and consequently the opportunity cost for reserves in our study did not differ from that of satellite-identified areas.

The objective of the forest owner selecting retention patches might be different from that of the Swedish Forest Agency and the County Administrative Board selecting key habitats and nature reserves. Whereas key habitats and reserves are selected to maximize conservation value, retention patches may be selected to minimize costs while still complying with legal and certification requirements. In addition, the planning area for retention patches is restricted to a single stand, which limits the possibilities for efficient conservation planning (Strange et al., 2006). In our study, retention patches had low amounts of large-diameter trees per ha, indicating either the removal of such trees, areas of lower productivity, or younger parts of the stand. Hence, the opportunity cost per area was considerably lower than in the other strategies. However, as the retention requirements are based on volume of trees rather than area, this should have little influence on the forest owners' opportunity costs. The density of biodiversity-related structures as well as of species was low in retention patches but, because of the low opportunity costs, retention patches were the second most cost-effective strategy. Nevertheless, although much fewer plots could be protected at a certain budget level with the key habitat strategy, the biodiversity value of those plots would be higher than in the larger number of plots possible to retain with the retention patch strategy.

We found that the information costs associated with the different strategies correlated well to the importance of conservation value in the selection process. For key habitats high conservation value is the only important parameter and the information cost for this strategy was also the highest. As nature reserves have to be large the need for detailed information about conservation value is lower and so was the information cost associated with the strategy. In the retention patch strategy the importance of conservation value is likely to be even lower, because of the limited selection possibilities and the possible importance of other parameters. Accordingly, we found the information cost in this strategy to be the lowest among the implemented strategies. In fact, it is in the interest of forest owners to keep information costs down as long as information about conservation value does not bring additional benefits (Hanley et al., 1997). This might be more pronounced for small private forest owners than for larger forest companies, since the latter have a greater potential to directly communicate the "biodiversity quality" of their production to the market. This may at least partially explain why the compliance with the law is lower for small forest owners than for large forest companies (Swedish Forest Agency, 2007a).

In general, information costs were low compared to opportunity costs for all strategies and they did not influence the result of the cost-effectiveness analysis. Therefore, we can conclude that the investment in information paid off for all

implemented strategies as they were more cost-effective than the random selection of old spruce forests, also after information costs had been included. It is even likely that the cost-effectiveness of the strategies would increase if more information, both about conservation value and especially about opportunity costs, would be collected (Drechsler and Wätzold, 2001; Naidoo et al., 2006; Murdoch et al., 2007).

Different biodiversity-related structures were most cost-effectively included by different conservation strategies and in this respect they were therefore complementary. Contrary, there was little variation between species groups in terms of which strategy was the most cost-effective. In fact, all groups of red-listed species were most cost-effectively included by the key habitat strategy. This means that there was little evidence of complementarity for different species groups between strategies. Neither did we find evidence of complementarity of strategies within species groups, as the cost-effectiveness of combinations of strategies was generally intermediate compared to that of pure strategies. Hence, if areas selected with different strategies tended to contain different species within a species group, these differences were not large enough to outweigh the disadvantage of combining a more cost-effective strategy with a less cost-effective one. Therefore, with respect to our study species, cost-effectiveness could not be increased by combining strategies. Nevertheless, strategies might be complementary with respect to other species groups. Furthermore, strategies could be complementary in terms of who carries the opportunity costs. In this case, the opportunity costs that the forest owners carry for retention patches can perhaps not be relocated to key habitats as forest owners may be more willing to contribute to conservation if they can influence the location of conservation areas (Pouta et al., 2002).

The intention of our study was that the results should reflect the differences between the strategies in terms of how conservation areas were selected. However, a general problem with implemented strategies is that they may differ in several factors and it can be difficult to separate the effects that the different factors have on the cost-effectiveness of a strategy. For example, the small size of retention patches and the fact that in our study they were always surrounded by clear-cut area means that our sample plots (although placed in the centre) were influenced by edge effects. Although this is a direct effect of the strategy it is not associated with the identification and selection of conservation areas.

Over a longer time period, differences other than the selection process may become more important for the cost-effectiveness of the strategies. For example, key habitats are generally small, which makes them vulnerable to edge effects and risks associated with small populations (Hanski, 2000; Aune et al., 2005). However, it is important to maintain habitats where species important to conservation are known to exist. It should therefore be investigated whether buffer zones around key habitats could be a cost-effective complement to this strategy. Retention patches are even smaller than key habitats and the changes in habitat quality and species content over time are expected to be large. However, one intention with the retention patch strategy is to increase the amounts of structures important to biodiversity conservation in the developing stand (Franklin et al., 1997; Rosenvald

and Löhmus, 2008) and therefore the value of this measure may increase with time. Nature reserves are on the other hand large and may be able to maintain populations of species requiring large and relatively undisturbed areas.

The results of our study are specific for the species groups and the forest type studied as well as for the type of landscape in which it was conducted. However, beetles, bryophytes, and lichens are key components of this ecosystem. They make up a large proportion of both the species richness and the biomass (e.g. Pettersson et al., 1995; Esseen et al., 1996) and almost 50% of the red-listed forest species in the area (Gärdenfors, 2005). They also play major roles in the nutrient cycling (e.g. Wilding et al., 1989; DeLuca et al., 2002). In addition, the relative cost-effectiveness of the strategies was similar for all three species groups. We therefore believe that our results well represent the effects that different selection processes have on the cost-effectiveness of including species in conservation areas. With respect to forest type, old spruce forest has high opportunity costs compared to information costs. In forest types with lower economic value the information costs would be relatively more important and expensive inventories would be less likely to pay off. With respect to landscape, middle boreal Sweden has a long history of intensive forestry (Andersson and Östlund, 2004) and old-growth forests occur only as scattered fragments. This makes the selection of large conservation areas less cost-effective, an effect which will be smaller in less fragmented landscapes (Ranius and Kindvall, 2006). If different strategies are most cost-effective in different types of forests or in different landscapes applying the right strategy in the right context becomes very important.

For conservation strategies in general, the cost-effectiveness in terms of selecting conservation areas depends on the size of the planning area, the size of the conservation areas, the objective of the agent making the selection, and the amount and type of information on which the selection is based. It will also depend on in which landscape, to which forest type and in which social context it is applied. The cost-effectiveness could also be different for different biodiversity components. Furthermore, there are trade-offs between the selection process and other parts of a conservation strategy. For example, a strategy requiring large conservation areas will be less cost-effective in selecting areas with high conservation value but may be more cost-effective for long-term survival of species. Given all these variables it is currently difficult to predict which type of strategy will be the most cost-effective given certain conditions. We therefore propose to both develop a theoretical framework for cost-effectiveness of conservation strategies and to increase the number of studies of actually implemented strategies. This work will require the input of many disciplines, including ecology, economics, forest management, and also sociology and psychology.

Acknowledgements

We thank Brita Svensson, Richard Hopkins, and two anonymous reviewers for valuable comments on the manuscript. The study was financed by the Swedish Research Council

for Environment, Agricultural Sciences and Spatial Planning (Formas), and the Forestry Research Institute of Sweden (Skogforsk).

REFERENCES

- Andersson, R., Östlund, L., 2004. Spatial patterns, density changes and implications on biodiversity for old trees in the boreal landscape of northern Sweden. *Biological Conservation* 118, 443–453.
- Ando, A., Camm, J., Polasky, A.S., 1998. Species distributions, land values, and efficient conservation. *Science* 279, 2126–2128.
- Angelstam, P., Mikusinski, G., Eriksson, J.A., Jaxgård, P., Kellner, O., Koffman, A., Ranneby, B., Roberge, J.M., Rosengren, M., Rystedt, S., Rönnbäck, B.-I., Seibert, J., 2003. Gap Analysis and Planning of Habitat Networks for the Maintenance of Boreal Forest Biodiversity – a Technical Report from the RESE Case Study in the Counties of Dalarna and Gävleborg. Department of Natural Sciences, Örebro University and Department of Conservation Biology, Swedish University of Agricultural Sciences.
- Anon., 2000. Svensk FSC-standard för certifiering av skogsbruk, second ed. Svenska FSC-rådet, Uppsala (in Swedish).
- Aune, K., Jonsson, B.G., Moen, J., 2005. Isolation and edge effects among woodland key habitats in Sweden: is forest policy promoting fragmentation? *Biological Conservation* 124, 89–95.
- Berg, A., Ehnström, B., Gustafsson, L., Hallingbäck, T., Jonsell, M., Weslien, J., 1994. Threatened plant, animal, and fungus species in Swedish forests – distribution and habitat associations. *Conservation Biology* 8, 718–731.
- Cabeza, M., Moilanen, A., 2001. Design of reserve networks and the persistence of biodiversity. *Trends in Ecology & Evolution* 16, 242–248.
- Colwell, R.K., 2006. Estimates: Statistical Estimation of Species Richness and Shared Species from Samples. Version 8. Persistent <purl.oclc.org/estimates>.
- Colwell, R.K., Mao, C.X., Chang, J., 2004. Interpolating, extrapolating, and comparing incidence-based species accumulation curves. *Ecology* 85, 2717–2727.
- Convention on Biological Diversity, 1992. United Nations Conference on Environment and Development, Rio de Janeiro. <www.biodiv.org>.
- DeLuca, T.H., Zackrisson, O., Nilsson, M.-C., Sellstedt, A., 2002. Quantifying nitrogen-fixation in feather moss carpets of boreal forests. *Nature* 419, 917–920.
- Djupström, L.B., Weslien, J., Schroeder, L.M., 2008. Dead wood and saproxylic beetles in set-aside and non set-aside forests in a boreal region. *Forest Ecology and Management* 255, 3340–3350.
- Drechsler, M., Wätzold, F., 2001. The importance of economic costs in the development of guidelines for spatial conservation management. *Biological Conservation* 97, 51–59.
- Ekö, P.M., 1985. A Growth Simulator for Swedish Forests, Based on Data from the National Forest Survey. Department of Silviculture, Swedish University of Agricultural Sciences, Umeå (in Swedish, English summary).
- Ekvall, H., 2001. Plan 33 – Ett verktyg för ekonomisk analys av skogsbruksföretagets virkesproduktion. Licentiate Thesis, Report No. 123. Department of Forest Economics, Swedish University of Agricultural Sciences, Umeå (in Swedish, English abstract).
- Esseen, P.A., Renhorn, K.E., Petersson, R.B., 1996. Epiphytic lichen biomass in managed and old-growth boreal forests: effect of branch quality. *Ecological Applications* 6, 228–238.
- Faustmann, M., 1849. Berechnung des Wertes welchen Waldboden sowie noch nicht haubare Holzbestände für die

- Waldwirtschaft besitzen. Allgemeine Forst- und Jagtzeitung, vol. 15, pp. 441–455 (English translation in Gane, M. (Eds.), Linnard, W. (transl.), Martin Faustmann and the Evolution of Discounted Cash Flow: Two Articles from Original German of 1849. Commonwealth Forestry Institute, University of Oxford, Institute Paper 42, 1968. Republished with permission in: Journal of Forest Economics 1, 7–44. Reprinted 1995).
- Fazey, I., Fischer, J., Lindenmayer, D.B., 2005. What do conservation biologists publish? *Biological Conservation* 124, 63–73.
- Ferraro, P.J., Pattanayak, S.K., 2006. Money for nothing? a call for empirical evaluation of biodiversity conservation investments. *Plos Biology* 4, 482–488.
- Franklin, J.F., Berg, D.R., Thornburgh, D.A., Tappeiner, J.C., 1997. Alternative silvicultural approaches to timber harvesting: variable retention harvest systems. In: Kohm, K.A., Franklin, J.F. (Eds.), *Creating a Forestry for the 21st Century: The Science of Ecosystem Management*. Island Press, Washington, DC, pp. 111–139.
- Freitag, S., Van Jaarsveld, A.S., 1998. Sensitivity of selection procedures for priority conservation areas to survey extent, survey intensity and taxonomic knowledge. *Proceedings of the Royal Society in London B* 265, 1475–1482.
- Gärdenfors, U. (Ed.), 2005. The 2005 Red List of Swedish species. Swedish Species Information Centre, Swedish University of Agricultural Sciences, Uppsala.
- Grand, J., Cummings, M.P., Rebelo, T.G., Ricketts, T.H., Neel, M.C., 2007. Biased data reduce efficiency and effectiveness of conservation reserve networks. *Ecology Letters* 10, 364–374.
- Groeneveld, R., 2005. Economic considerations in the optimal size and number of reserve sites. *Ecological Economics* 52, 219–228.
- Hägglund, B., 1979. Ett system för bonitering av skogsmark: analys, kontroll och diskussion inför praktisk tillämpning. Report No. 14, Projekt HUGIN. Swedish University of Agricultural Sciences, Umeå (in Swedish).
- Hägglund, B., Lundmark, J.-E., 1981. Handledning i bonitering med Skogshögskolans boniteringsystem Del 1,2,3 Skogsstyrelsen, Jönköping (in Swedish).
- Hanley, N., Shogren, J.F., White, B., 1997. *Environmental Economics in Theory and Practice*. Macmillan, London.
- Hanski, I., 2000. Extinction debt and species credit in boreal forests: modelling the consequences of different approaches to biodiversity conservation. *Annales Zoologici Fennici* 37, 271–280.
- Kolstad, C.D., 2000. *Environmental Economics*. Oxford University Press, New York.
- Lindenmayer, D.B., Franklin, J.F., Fischer, J., 2006. General management principles and a checklist of strategies to guide forest biodiversity conservation. *Biological Conservation* 131, 433–445.
- Margules, C.R., Pressey, R.L., 2000. Systematic conservation planning. *Nature* 405, 243–253.
- McCann, L., Colby, B., Easter, K.W., Kasterine, A., Kuperan, K.V., 2005. Transaction cost measurement for evaluating environmental policies. *Ecological Economics* 52, 527–542.
- Merlo, M., Paveri, M., 1997. Formation and implementation of forest policies: a focus on the policy tool mix. In: *Proceedings of XI World Forestry Congress*. Antalya, Turkey, 13–22 October 1997, vol. 5, pp. 233–254.
- Millennium Ecosystem Assessment, 2005. *Ecosystems and Human Well-being: Policy Responses*. Washington, DC.
- Murdoch, W., Polasky, S., Wilson, K.A., Possingham, H.P., Kareiva, P., Shaw, R., 2007. Maximizing return on investment in conservation. *Biological Conservation* 139, 375–388.
- Naidoo, R., Adamowicz, W.L., 2005. Economic benefits of biodiversity exceed costs of conservation at an African rainforest reserve. *Proceedings of the National Academy of Sciences of the United States of America* 102, 16712–16716.
- Naidoo, R., Balmford, A., Ferraro, P.J., Polasky, S., Ricketts, T.H., Rouget, M., 2006. Integrating economic costs into conservation planning. *Trends in Ecology & Evolution* 21, 681–687.
- Nitare, J., Norén, M., 1992. Woodland key habitats of rare and endangered species will be mapped in a new project of the Swedish National Board of Forestry. *Svensk Botanisk Tidskrift* 86, 219–226 (in Swedish, English summary).
- Norén, M., Nitare, J., Larsson, L., Hultgren, S., Bergengren, I., 2002. *Handbok för inventering av nyckelbiotoper*. Swedish Forest Agency, Jönköping (in Swedish).
- Payton, M.E., Greenstone, M.H., Schenker, N., 2003. Overlapping confidence intervals or standard error intervals: what do they mean in terms of statistical significance? *Journal of Insect Science* 3, 1–6.
- Perhans, K., Gustafsson, L., Jonsson, F., Nordin, U., Weibull, H., 2007. Bryophytes and lichens in different types of forest set-asides in boreal Sweden. *Forest Ecology and Management* 242, 374–390.
- Pettersson, R.B., Ball, J.P., Renhorn, K.E., Esseen, P.A., Sjöberg, K., 1995. Invertebrate communities in boreal forest canopies as influenced by forestry and lichens with implications for passerine birds. *Biological Conservation* 74, 57–63.
- Pouta, E., Rekola, M., Kuuluvainen, J., Li, C.Z., Tahvonen, I., 2002. Willingness to pay in different policy-planning methods: insights into respondents' decision-making processes. *Ecological Economics* 40, 295–311.
- Ranius, T., Kindvall, O., 2006. Extinction risk of wood-living model species in forest landscapes as related to forest history and conservation strategy. *Landscape Ecology* 21, 687–698.
- Ranius, T., Ekvall, H., Jonsson, M., Bostedt, G., 2005. Cost-efficiency of measures to increase the amount of coarse woody debris in managed Norway spruce forests. *Forest Ecology and Management* 206, 119–133.
- Richardson, E.A., Kaiser, M.J., Edwards-Jones, G., Possingham, H.P., 2006. Sensitivity of marine-reserve design to the spatial resolution of socioeconomic data. *Conservation Biology* 20, 1191–1202.
- Rosenvald, R., Löhmus, A., 2008. For what, when, and where is green-tree retention better than clear-cutting? a review of the biodiversity aspects. *Forest Ecology and Management* 255, 1–15.
- Sala, O.E., Chapin, F.S., Armesto, J.J., Berlow, E., Bloomfield, J., Dirzo, R., Huber-Sanwald, E., Huenneke, L.F., Jackson, R.B., Kinzig, A., Leemans, R., Lodge, D.M., Mooney, H.A., Oesterheld, M., Poff, N.L., Sykes, M.T., Walker, B.H., Walker, M., Wall, D.H., 2000. Biodiversity – global biodiversity scenarios for the year 2100. *Science* 287, 1770–1774.
- Saterson, K.A., Christensen, N.L., Jackson, R.B., Kramer, R.A., Pimm, S.L., Smith, M.D., Wiener, J.B., 2004. Disconnects in evaluating the relative effectiveness of conservation strategies. *Conservation Biology* 18, 597–599.
- Schenker, N., Gentleman, J.F., 2001. On judging the significance of differences by examining the overlap between confidence intervals. *The American Statistician* 55, 182–186.
- SKSFS, 1993. SKSFS 1993:2. Skogsstyrelsens föreskrifter och allmänna råd till skogsvårdslagen (SFS 1979:429) (in Swedish). StatSoft, Inc., 2005. *Statistica (Data Analysis Software System)*. Version 7.1. <www.statsoft.com>.
- Strange, N., Rahbek, C., Jepsen, J.K., Lund, M.P., 2006. Using farmland prices to evaluate cost-efficiency of national versus regional reserve selection in Denmark. *Biological Conservation* 128, 455–466.
- Sutherland, W.J., Pullin, A.S., Dolman, P.M., Knight, T.M., 2004. The need for evidence-based conservation. *Trends in Ecology & Evolution* 19, 305–308.
- Swedish Environmental Protection Agency, Swedish Forest Agency, 2005. *National Strategy for the Formal Protection of Forests*. Naturvårdsverket och Skogsstyrelsen (in Swedish, English summary).

- Swedish Forest Agency, 2007a. Statistical Yearbook of Forestry 2007. Swedish Forest Agency, Jönköping (in Swedish with English summary).
- Swedish Forest Agency, 2007b. Fördjupad utvärdering av Levande skogar. Meddelande 4. Skogsstyrelsen, Jönköping (in Swedish).
- Swedish Forest Agency, 2008. <<http://www.svo.se/episerver4/templates/SNormalPage.aspx?id=13083>> (accessed 16.04.08).
- Wätzold, F., Schwerdtner, K., 2005. Why be wasteful when preserving a valuable resource? a review article on the cost-effectiveness of European biodiversity conservation policy. *Biological Conservation* 123, 327–338.
- Wilding, N., Collins, N.M., Hammond, P.M., Webber, J.F. (Eds.), . *Insect-Fungus Interactions*. Academic Press, London.