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Excepted coverage of endangered species on Danish heathland

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Abstract:
Considerable resources must be allocated for fulfilling the Habitat Directive and the question of optimal allocation is as important as it is difficult. In the present study, we estimate the expected species coverage of three non-probabilistic strategies: i) a maximum selected area strategy, ii) a hotspot selection strategy, and iii) a minimising cost strategy, and one probabilistic strategy a maximum expected coverage strategy. We show that the optimal network changes considerably with strategies. Thus, the study provides insights which may guide conservation authorities on how to target their actions so that they accomplish the most with limited budgets, while acknowledging the uncertainty of species presence. We finally discuss how welfare economic evaluations of conservation targets could be included in such conservation policy analysis.

Keywords**:** biodiversity, expected coverage, heathland protection

1 Introduction

1.1 Planning for conservation

Pressures on natural habitats from agriculture, accelerating urbanisation, pollution and tourism have caused massive destruction of habitats and led to a marked decline in species populations with the result that half of Europe's mammal species and one third of reptiles, fish and bird species are currently endangered (European Commision 1997). According to the EU Habitat Directive, member states are obliged to establish a network of protected areas, known as Natura 2000, where special actions need to be taken to conserve biological diversity. Natura 2000 plans must be implemented before 2009, and actions taken to ensure that appropriate conservation status of the habitats is maintained. Considerable resources must be allocated for fulfilling these requirements, knowing that species presence may be uncertain, and connected with the stochastic nature of environment as well as area treatment and management. This increases the interest in identifying which areas of a given habitat are the most important to protect in order to conserve biological diversity in a probabilistic context. Hence, conservation authorities face the problem of designing their actions so as to accomplish the most with limited budgets, while acknowledging the uncertainty of species presence. Political priority settings require that the network design of reserved sites relies on sound biological information, not only information on species occurrence but also viability measures. Fragmentation of habitats may often lead to the formation of meta-populations (Hanski and Gilpin 1991), which may be more sensitive to local extinction if the species can not colonise new sites (Harrison 1994).

Two methods have been proposed in the literature for solving the reserve site selection problem with probabilistic presence-absence species information. One is the 'expected coverage' approach, which maximises the expected number of species covered (Polasky and others 2000). The other method is the 'threshold approach', which maximises the number of species covered, where a species counts as covered only if the probability of coverage reaches a specified threshold (e.g., Margules and Stein 1989, Haight and others 2000). Both selection approaches find that reserve network sites differ significantly when using probabilistic data to maximise the expected number of species represented versus using deterministic approaches. Arthur and others (2002) compared the expected coverage approach and the threshold approach and found that information on habitat quality and species viability is important when designing the network. The expected coverage problem is formulated as a nonlinear binary integer programme, which belongs to the group of non-polynomial hard problems (Camm and others 2002). Since the individual probabilities are nonlinear functions of the decision variables the objective functions cannot be transformed into linear form. However, Camm and others (2002) and Arthur and others (2004) show that linear approximations of the nonlinear problem can yield good solutions to even large problems. The present study builds on this linearisation procedure.

1.2. Study area - the Danish heathland

One of the natural habitats covered by the Natura 2000 regulations is the heathlands which are selected as case object for this study. The Habitat and Wild Birds Directives have been implemented in the Danish Forest Law, Nature Protection Law and Law of Environmental Objectives. The articles state that the EU member states shall establish the necessary conservation measures involving, if need be, appropriate management plans specifically designed for the sites or integrated into other development plans. Hence the member states need to take steps in order to maintain or restore the natural habitat types and species at a favourable conservation status.

The origin of the Danish heath can be traced back to the over-exploitation of poor soils since the beginning of the Bronze-Age and covered more than 600,000 hectares by year 1822 (Hansen, 1970). Today, the Danish heathland is mainly located in the western and northern parts of Jutland and cover roughly 80,000 hectares, or approximately 2% of the total land area (Buttenschön, 1993). The drastic reduction in area is largely attributed to cultivation of the heath (Hansen, 1970). Today, heath areas are protected by law from being converted into other uses. Nevertheless, atmospheric nitrogen deposition and lack of the nutrient-removing traditional agricultural practices are allowing grasses, bushes and trees to take over. The natural processes of nitrogen deposition are currently being accelerated by nitrogen being deposited from nearby farms and traffic. The nutrient-poor heath has a special flora and fauna which is not found elsewhere in Denmark. Twenty-five species unique for the heath in Denmark are red-listed as either critically endangered, endangered or vulnerable (Stoltze and Pihl, 1998), but it should be noted that all species also exist outside Denmark. Furthermore, the heath has a cultural value as a landscape type, e.g. described in the national romantic literature and art (e.g. Raadal 1942, Andersen 1860, Blicher 1920-34). The results of a brief telephone survey among responsible regional and state authorities suggests that currently about one fourth of the area is managed such as to preserve the heath ecosystem; the remainder is slowly being overgrown.

2. Methods and Materials

2.1 Biodiversity data

We use a geographically distributed data set on terrestrial and freshwater species to identify the potentially endangered occurrence of species on Danish heathland. Information on species distribution (and species assemblages within each cell in a grid) is compiled as present/absent, based on all Danish summer atlas data providing complete coverage of all species within a given taxon. The data set species include 41 orchids (Orchidaceae; Wind 2001), 18 species of crawling water beetles (Coleoptera: species within Haliplidae; Holmen 1981), 23 species of click beetles (Coleoptera: species within Elateridae; Martin 1989), 41 goldsmiths (Nielsen 1998), 26 grasshoppers (Nielsen 2000), 61 species of butterflies (Lepidoptera: species within Hesperioidea, Papilionoidea; Stoltze 1994), 156 species of large moths (Lepidoptera: species within Hepialoidea, Cossoidea, Zygaenoidea, Tineoidea, Yponomentoidea, Bombycoidea, Geometroidea, Sphingoidea, Notodontoidea, Noctuoidea;

Kaaber 1982), 252 species of hoverflies (Diptera: species within Syrphidae; Torp 1994), 19 species of amphibians and reptiles (Amphibia/Reptilia; Fog 1993), 48 mammals, 6 Lycopodium complanatum (Pihl and others 2000), 189 species of birds (Aves; Stoltze 1994). The database thus comprises a total of 1006 species of the "estimated" 30,000 species in Denmark (Stoltze and Pihl 1998, see Petersen and others 2005 for use and further description of this database). The data base contains one of the most complete data sets on natural flora and fauna species in Europe.

We generated a list of 11 endangered red-listed species from the total data set, which all are related to the heath habitat, i.e. the database does not contain accurate distribution data on the occurrence of all the above-mentioned 25 endangered species. The species include 2 hoverflies, 3 grasshoppers, and 6 large moths (see Table 1). The data are located in the 633 UTM (Universal Transverse Mercator) 10 x 10 km grid cells which provide a complete coverage of Denmark. For each of the 633 UTM cells we calculated the actual amount of heath land using data from The Danish Area Information System, which contains 40 detailed data layers, with area information based on more than two million polygons and with a precision of +/- 25 meters (Danish Ministry of Environment and Energy, 2000).

The geographical representation of the 11 red listed species is presented in Figure 1a and heath land areas in Figure 1b, both in the 10 x 10 km grid cells.

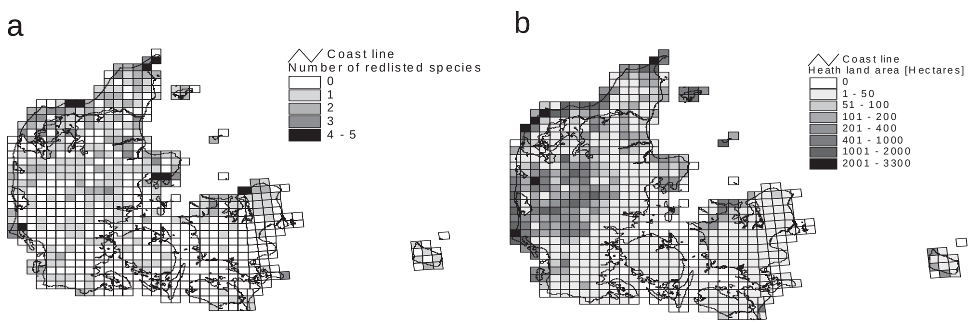

Figure 1. Red-listed species distribution and stratification of heath areas in Denmark using a spatial grain size of 10 x 10 km grid cells. (a) Species density of the 11 species included in the study, (b) heathland density.

2.2 Calculation of persistence probabilities

Species-specific persistence probabilities were calculated for all 11 species, based on the assumption that local persistence depends on the current abundance of species inside the grid cell, the habitat quality and the size of heath land. The abundance of the 11 redlist species inside each grid cell is based on the complete data base of species. If the species is not abundant in the grid cell, we assume the probability of persistence is zero. Otherwise the persistence probability is a composite of pressure on habitat quality and size. A major threat against habitat quality of Danish heathland is increasing air pollution loads caused by airborne nitrogen and phosphor from farms, traffic, energy production and industry

(European Environment Agency 2003). The aim of the Danish EUDANA-project is to develop and adjust existing tools for calculation of critical loads based on biodiversity targets and threshold values for favourable conservation status of terrestrial nature types, including heath, protected by the EU Habitat Directive and by the Danish Nature Protection Law (Bak and Ejrnæs 2004). We use Bak (2001) and Bak and Ejrnæs (2004) to assess the share of heath areas which have a critical load above the threshold values. In the following, we assume that shares of [0.05, 0.15, 0.25, 0.35, 0.45, 0.55, 0.65, 0.75, 0.85, 0.95] result in the following probabilities, λ_j, of a habitat, j, becoming degraded [0.50, 0.55, 0.61, 0.66, 0.72, 0.77, 0.83, 0.88, 0.94, 0.99], provided it is part of the conservation network. Otherwise, we assume the habitat will for sure be degraded (see the distribution in Figure 2).

However, species may be more or less sensitive to degradation of their habitat. Table 1 shows the 11 red listed species, their representation and assumed probabilities, φ_i for species i, of becoming locally extinct given habitat degradation using the IUCN criteria for 'Critically Endangered', 'Endangered' and 'Vulnerable' species (The Danish Red Data Book, 2005).

Table 1. Species list, red-list category and probability of extinction. The last column displays the number of grid cells, cf. Figure 1a, where the species is present.

Species name, i	Family	Red List Category	Probability of extinction given degradation, φ_i,%	Species representation
Chamaesyrphus lusitanicus	Hoverflies	Vulnerable	20	8
Chortippus mollis	Grasshoppers	Vulnerable	20	1
Erynnis tages	Moth	Endangered	40	48
Euphydryas aurinia	Moth	Critically endangered	100	6
Maculinea alcon	Moth	Vulnerable	20	3
Maculinea arion	Moth	Critically endangered	100	3
Oedipoda caerulescens	Grasshoppers	Critically endangered	100	1
Omocestus haemorrhoidalis	Grasshoppers	Critically endangered	100	5
Paragus finitimus	Hoverflies	Vulnerable	20	24
Plebejus argus	Moth	Vulnerable	20	120
Pyrgus malvae	Moth	Vulnerable	20	193

We assume that local extinction in a grid cell belonging to the network will only take place provided the habitats in that cell are degraded first. That is, for non-degraded habitats, the extinction probability is set to zero. For sites not in the network, however, the species is assumed to go extinct with certainty. Furthermore, we follow Gaston and others (2002) who show that the probability measures of local extinction may improve if we include a measure of habitat size. The resulting model for the persistence probability is defined as:

$$p_{ij} = \begin{cases} (1-\lambda_j\varphi_i)\cdot(0.0705\cdot Ln(\text{Area}_j+0.01)+0.4247) & if \quad j \in J_i \\ 0 \quad otherwise \end{cases} \qquad (1)$$

, where λ_j is the probability of habitat degradation as given in the vector above , φ_i is the probability of species i becoming locally extinct if the habitat is degraded, cf. Table 1, and $Ln(\text{Area}_j+0.01)+0.4247$ is a natural logarithmic adjustment function, which is 1 when habitat j is larger than 3500 hectares and 0.1 when the area is close to zero. If species i is not present in site $j \in J_i$, the persistence probability is set at zero. Hence, for large areas (>3500 hectares) there is no adjustment of persistence probability, the opposite holds for smaller areas.

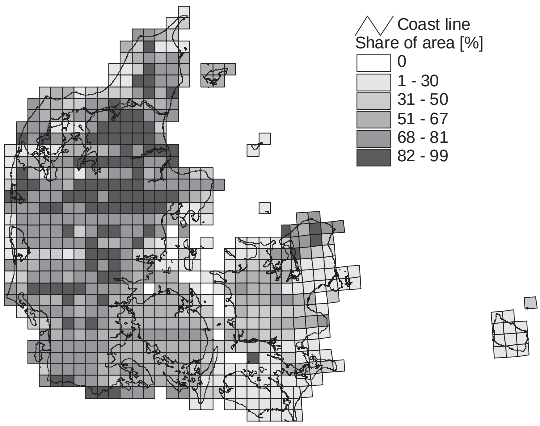

Figure 2. The share of heathland located in the grid cells which have a critical load above the threshold values.

2.3 Management and cost calculation

Heath lands have mainly been created and maintained by former agricultural management practices involving regular export of nutrients from the actual site (Webb 1998), e.g. by management in the form of sod-cutting, controlled burning and grazing. The disappearance of traditional management in line with increasing airborne nutrient deposits has enforced the heath land decline over the past few decades (Marrs 1993, Rose and others 2000). The precise form, and frequency, of habitat management will affect not only the regeneration potential of heath plant communities, but also the extent of nutrient removal from the system. Finally, this response of plant communities will exert an influence on the persistence possibilities of other species.

Controlled burning (Hobbs and Gimingham 1980, Bullock and Webb 1995) can remove as much as 95% of the nitrogen in the above ground portion of the plant communities (Chapman 1967). Other systems like grazing may also remove considerable amounts of nitrogen (Bullock and Pakeman 1997). However, the above-ground biomass accounts for less than 20% of the total nitrogen stores (Power and others 1998). Hence, managements that also remove the litter (e.g. Allison and Austen 2006) and/or humus layers, like sod/turf cutting (De Graaf and others 1998, Britton and others 2000) will result in a more substantial decrease in organic nitrogen stores in heathland ecosystems, thereby improving the conditions for *Calluna vulgaris*-dominated heathland vegetation.

Most heathland in Denmark is maintained by the Danish Forest and Nature Agency and most of it is managed by grazing and cutting the turf. The annual cost of heathland management is data supplied by the State Forest Districts (Danish Forest and

Nature Agency 2004). The cost data reveal that economics of scale prevail, since the cost per hectare decreases with increasing size of the maintained area. The cost of managing and operating on 100 hectares is EUR 78.4 per hectare, whereas the cost of managing 7000 hectares is approximately EUR 26.7 per hectare. The distribution of management cost is shown in Figure 3.

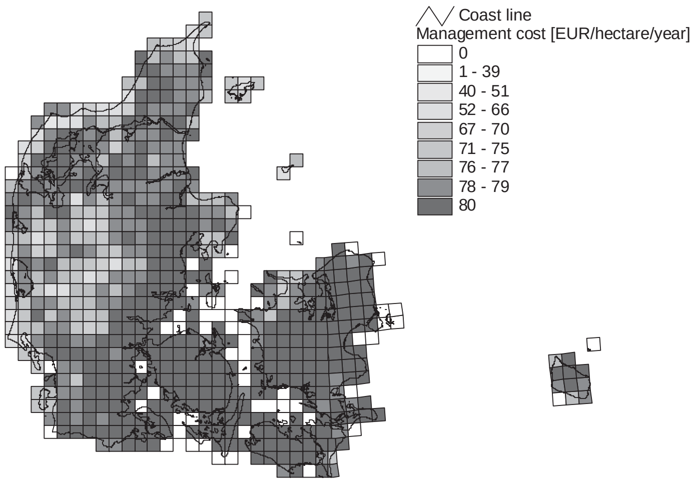

Figure 3. Distribution of heathland management cost

2.4 Non-probabilistic conservation policies

Three non-probabilistic conservation strategies were evaluated in the present study:

- Maximise selection area: This strategy implies that the main goal is to identify the maximum reserve area within a budget constraint, ignoring all species information.
- Hotspot selection: We use *a priori* information on the 11 red-listed species occurrences and apply a hotspot strategy where the most species-rich sites are selected (Myers 1988, Prendergast and others 1993). This corresponds to sorting the grid cells according to the number of species identified, and selecting the cells containing the highest number of species until the budget limit has been exceeded.
- Minimum cost representation: Since the hot spot strategy does not guarantee that all species are covered in the network, we apply the minimum cost representation strategy (Margules and others 1988), which ensures that all species are represented in the network at the lowest cost. Let *I* represent the set of threatened or endangered species in the database and let $X_j = 1$ if species *i* is present in site *j*, zero otherwise. Species *i* is present in all sites $j \in J_i$. The management cost is denoted c_j. Then the

minimum cost representation is: Minimise the opportunity costs, $\sum_{j \in J} c_j X_j$ while, fulfilling the requirement that all 11 endangered red-listed species should at least be represented r_i times in the network, $\sum_{j \in J_i} X_j \geq r_i$, or at least the maximum number of times the species occurs if r_i exceeds this number, $r_i = \#\{j \mid j \in J_i\}$, for all $i \in I$.

2.5 The expected coverage model

Species presence may be uncertain and related to the stochastic nature of habitat quality as well as management. Following the notion above, let p_{ij}, be the probability that species $i \in I$ persists at site $j \in J$. We assume that $0 \leq p_{ij} < 1$, cf. Equation 1. Assuming that probabilities are independent between sites the probability that a species does not persist in any of the sites, s_i, can be estimated as $s_i = \prod_{i \in I} (1 - p_{ij})^{X_j} \quad \forall \quad i \in I$.

Red-listed species may require particular attention and minimum coverage probabilities. Here we introduce the minimum probability threshold, h_i, that species i must be included in the selected sites at a minimum probability level $(1 - s_i) \geq h_i$. We set the minimum probability level at 0.4 for all 11 species.

The problem is nonlinear and we use the procedure of Arthur and others (2004) to create the linear approximation for ln(s_i). A set of K breakpoints is applied to approximate the interval L to 1, where $L > 0$ is the lowest possible probability of species persistence, if as many sites as possible within the budget limit were included in the network. Then the probability that a species is not covered, s_i, can be estimated as $\sum_{k=1}^{K} B_k b_{ik}$ where $\sum_{k=1}^{K} b_{ik} = 1$, B_k is the kth breakpoint and b_{ik} is a continuous variable that weights the the kth breakpoint for species i. In a natural logarithmic transformation this can be formulated as $\sum_{k=1}^{K} \ln(B_k) b_{ik}$. The management cost per hectare is assumed to be decreasing with area size (see section 2.3). Hence, we approximate the estimated non-linear cost function using a piece-wise linear function (Eqs. 7 and 8). The final model is expressed as:

$Max \quad w$

Subject to:

$$\sum_{k=1}^{K} \ln(B_k) \cdot b_{ik} = \sum_{j \in J} X_j \ln(1 - p_{ij}) \qquad \forall \quad i \in I \quad (2)$$

$$\sum_{j \in J} X_j \ln(1 - p_{ij}) \leq \ln(1 - h_i) \qquad \forall \quad i \in I \quad (3)$$

$$s_i = \sum_{k=1}^{K} \ln(B_k) \cdot b_{ik} \qquad \forall \quad i \in I \quad (4)$$

$$w = \sum_{i \in I} 1 - s_i \qquad (5)$$

$$\sum_{j \in J} a_j X_j = Area \qquad (6)$$

$$\sum_{m=1}^{M} a_{jm} = a_j \qquad (7)$$

$$\sum_{m=1}^{M} z_m a_{jm} = c_j \qquad (8)$$

$$\sum_{j \in J} c_j X_j \leq Budget \qquad (9)$$

$$\sum_{k=1}^{K} b_{ik} = 1 \qquad \forall \quad i \in I \quad (10)$$

$$0 < p_{ij}, s_i, b_{ik} < 1$$

$$X_j \in \{0,1\}$$

, where w is the expected number of surviving species. a_j is the area of site j, a_{jm} and z_n are the coefficients of the piece-wise approximation of the management cost function in site j, and c_j is the resulting cost of including site j in the reserve network.

We simulated a number of scenarios changing number of breakpoints and level of breakpoints. The number of breakpoints was increased to provide a more accurate approximation of the non-linear expected coverage problem, but of course as noted by Arthur and others (2004), at the expense of more variables b_{ik}. We defined a set of 25 breakpoints to approximate the interval L to 1.0 and we set L=9.0 E-400. Experiments showed that the smaller intervals close to 0 and close to 1, the better approximations were achieved.

3. Results

The expected coverage values of the 11 red listed species of the two non-probabilistic conservation strategies (maximise selected heath area and hotspot) and the maximum expected coverage are presented for different budget levels in Figure 4. The lowest performance of the four strategies is found when using the area maximising algorithm. The second worst is the hotspot strategy, which however increases its performance from a rather poor level of 2.8 species to the fairly good level of 9.7 species at a budget constraint of EUR 4.01 million per year. The highest coverage is achieved using the maximum expected coverage approach, beginning at 4.7 species at an annual budget of EUR 0.058 million per year and ending with the maximum coverage of 9.7 species at budgets above EUR 4.01

million per year. Increasing budget levels seem not to increase expected coverage, which may indicate that welfare economic losses may occur if annual management cost are too high.

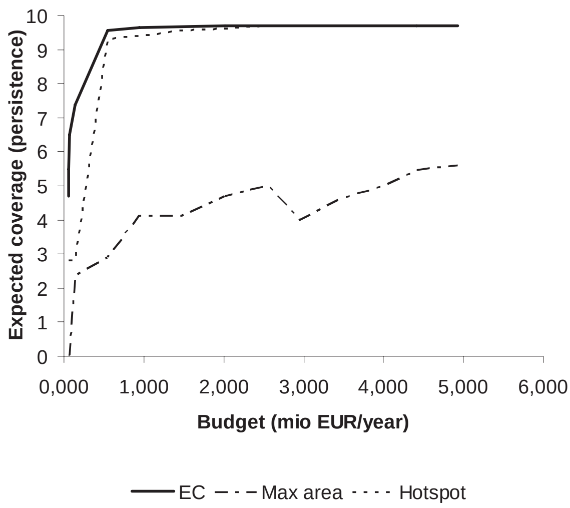

Figure 4. Expected coverage of the maximum expected coverage, maximum area and Hotspot strategies

We estimate the coverage of minimum set cost using a criterion of minimum 1 to 9 species representation. We find that the expected coverage increases only slowly when increasing the minimum required representations. It shows that the minimum set cost strategies do not take into account the probabilistic nature of the decision problem and that the minimum cost representation strategy is inferior to the other strategies, even the hot spot strategy.

4. Discussions

This study shows that conservation policies change significantly when including probabilistic measures of species persistence. This stresses the need for scientifically sound proxies and shows that the inclusion of information on biodiversity and estimates on species persistence can lead to a much more efficient protection of biodiversity in reserve networks (viz. when comparing to a non-probabilistic area maximising strategy). However, the result is ambiguous. We find that reserve selection strategies solely guided by area goals, aiming at protecting as large heath area as possible within budget limitations, out-perform the more traditional minimum set strategies. Hence, if species persistence information is not available, it may be more efficient to maximise the total heath area, rather than ensuring species representation in the network. Area goals will tend to favour large areas which, inherent to the model, are more capable to ensure the persistence of species located within their habitat. Minimum cost strategy will tend to prefer areas with lower persistence capabilities. Hence, our results supports the criticism against simple complementarity-based algorithms, which may fail to select areas where species have higher probabilities of persistence (e.g. Nicholls 1998, Williams 1998), thereby compromising the ultimate goal of efficient protection. Hot spot strategies favour areas with high species richness and higher persistence values, and hence will tend to perform better.

Another crucial feature of the model explored is the uncertainty of species survival. Persistence estimates are rarely available, and we chose to model an intuitively positive relation between species abundance and area size, and negatively related to ongoing environmental degradation. The larger the species abundance, the larger the sizes of areas and the lower the critical loads of air pollutants, the greater the probability that the species will survive. A number of studies attempt to develop models to estimate persistence (see e.g., Araujo and Williams 2000, Polasky and others 2000) either as probabilistic functions of suitable habitat, species currently colonising the area or threat/vulnerability. Some use expert opinions on species persistence within the areas (e.g., Arthur and others 2004). It is evident that the results are sensitive to these critical assumptions. However, since appropriate data for estimating species persistence are unavailable, we have to do with proxies.

Reserve site selection strategies and conservation management plans like those evaluated in this paper are rarely evaluated according to welfare economic measures. Rather a pure ecological approach is taken in most studies. This study shows that there may be a trade-off between benefits from expected species coverage/protection and conservation costs. We suggest that the presented modelling framework presented may be extended including benefit estimates of species protection (Boiesen et al. 2005) and an optimal level of management effort estimated.

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