Site selection for an umbrella species – white-backed woodpecker in Sweden - when costs and habitat quality are uncertain

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Working Paper Series 2010:6
Uppsala 2010

ISSN 1401-4068
ISRN SLU-EKON-WPS-10/6-SE

Abstract. This paper develops a stochastic dynamic model for the purpose of optimal site selection of habitats for an umbrella species in Sweden under conditions of uncertainty in the growth of habitat quality of established conservation areas and in acquisition costs. The numerical dynamic model builds on inputs from an ecological model of habitat development and an economic model of actual payments for biodiversity conservation in Swedish forests. The results point at the importance of including both types of uncertainties; total social costs for achieving given habitat targets under probabilistic constraints increase three fold as compared with the deterministic case. Another effect of the introduction of uncertainty is the earlier establishment of habitats due to need of extra establishments in order to achieve the target with a certain probability. When comparing optimal payment per ha conservation with actual payments it was noticed that there is a considerable difference among counties; while actual payments for some counties are quite close to optimal payments under any of the uncertainty conditions they can deviate largely for some other counties.

Key words: optimal habitat selection, biodiversity, uncertainty, spatial heterogeneity, numerical stochastic dynamic model, Sweden

1. Introduction

In spite of clear advantages of an ecosystem approach to conservation, single-species management is a major part of human attempts to halt biodiversity loss (Simberloff 1997). This strategy has been criticized for not being very effective and a more scientifically-based approach to the use of surrogate species in biodiversity conservation has been advocated (e.g. Caro and O'Doherty1999; Andelman and Fagan 2000; Favreau et al. 2006). Among different types of surrogate species used in conservation, the umbrella species concept appears to link single-species conservation with more community- or even ecosystem-oriented management delivering broader biodiversity benefits (Fleishman et al. 2000; Roberge and Angelstam 2004; Branton and Richardson 2010). The conservation of an umbrella species, usually a demanding specialist or species with large area requirements, is expected to guarantee that requirements of many co-occurring, less demanding species are also fulfilled.

Woodpeckers (Picidae) include several woodland birds that are sensitive to anthropogenic changes in forest environments (Mikusiński 2006). Clearing of forests and conversion of naturally dynamic forests to production landscapes have led to the drastic decline and sometimes extinctions of more specialized woodpeckers (e.g. ivory-billed or red-cockaded woodpeckers in North America). The incompatibility of woodpeckers with forestry is based on the fact that silviculture decreases areas with dead wood or big old trees that are crucial for woodpeckers. Due to this incompatibility several woodpecker species have been recognized as surrogates for the assessment of forest avian diversity and forest biodiversity in general (Mikusinski et al. 2001; Roberge et al. 2006, 2008a; Drever et al. 2008; Drever and Martin 2010).

Among European woodpeckers dependent on dead wood and deciduous trees, the white-backed woodpecker (*Dendrocopos leucotos*) currently receives a lot of attention among conservationists, forest managers and government agencies in Sweden (e.g., Mild and Stighäll 2005; Mikusiński et al. 2010). The Action Plan for the conservation of the white-backed woodpecker was approved by the Swedish Environmental Protection Agency (see Mild and

Stighäll 2005). The total budget amounted to SEK 200 million (approximately 21.5 million Euro) for the period 2005-2008 and funding continues. It is assumed that conservation measures directed at the white-backed woodpecker also will benefit many other species using the same habitat, i.e. this species would function as an 'umbrella species' for deciduous forest communities (Mild and Stighäll 2005). Several studies confirm the potential of this woodpecker to be an umbrella species (Martikainen et al. 1998; Roberge et al.2006, 2008b; Halme et al. 2009). The long-term objective of the action plan is to re-establish a favorable conservation status for the species in Sweden with over 200 breeding pairs by 2070.

However, given the time perspective of several decades both provision costs and the development of habitat quality are uncertain. Provision costs depend on opportunity cost of land and eventual management costs, such as creation of dead wood. Both these cost types are subjected to fluctuations caused by, among others, business cycles and economic growth in society. The development of habitat quality is affected by factors such as climate, ecological complexity and other environmental conditions which can not be predicted with certainty. The purpose of this study is to identify cost effective site selection of habitat establishment for the white-backed woodpecker in Sweden when considering uncertainty in both provision costs and the development of habitat quality in established sites. This is made by the construction of a stochastic dynamic model which builds on an ecological model of dynamic development of habitat quality in Swedish forests and on an economic study of actual costs for biodiversity conservation.

Starting in early 1980s, there is a considerable literature on site selection for biodiversity conservation (e.g. Kirkpatrick 1983; Margulles et al., 1988; Williams et al., 1996; Ando et al. 1998; Polasky et al., 2001; Wu and Skeleton 2002; Costello and Polasky 2004; Nalle et al., 2004; Newburn et al, 2006; Lewis et al., 2009). The literature is rooted in conservation biology which does not consider difference in conservation costs among different sites (e.g. Kirkpatrick 1983; Margulles et al., 1988; Williams et al., 1996). This is accounted for in the site selection literature based on economic theory (Ando et al. 1998; Polasky et al., 2001; Wu and Skeleton 2002; Costello and Polasky 2004; Nalle et al., 2004; Newburn et al, 2006). A common feature of most of the applied economics papers is that the quality of habitat at the reserve site is fixed. All papers consider spatial and dynamics factors affecting site selection, and a few also include uncertainty (e.g. Costello and Polasky, 2004; Langford et al. 2009).

The paper by Costello and Polasky (2004) is most similar to our paper with their focus on optimal dynamic site selection of biological reserves under conditions of uncertain development of habitat and constrained budget resources. However, to the best of our knowledge, there is no paper on optimal selection of conservation areas in time and space which accounts for uncertainty in both provision costs and habitat quality.

The paper is organized as follows. First we give a theoretical presentation of the stochastic dynamic model underlying the numerical calculations. Next we present parameterization of functions and data retrieval. Results with respect to optimal site selection are presented in Section 4, and policy analysis, with an evaluation of the Swedish conservation plan, are presented in Section 5. The paper ends with some tentative conclusions.

2. Model for optimal site selection

New habitats for our umbrella species (i.e. white-backed woodpecker, WBW) can be generated in $i=1,..,k$ different regions, and cover an area of P^i. We assume that without investment in WBW habitat improvement, the habitat quality of a given area remains constant, being determined by ("business-as-usual") timber production practices. If we create new reserves, however, habitat quality improves during time due to growth of deciduous trees, creation of deadwood etc., and reaches a maximum, steady state, level of quality. Due to differences in climatic and environmental conditions, K^i, both the initial quality and the maximum habitat quality of an established conservation area, $Q_0^i(P_0^i;K^i)$ and $Q^{iMax}(P^i;K^i)$ respectively, differ among regions. The time required for reaching $Q^{iMax}(P^i;K^i)$ depends on quality at the time of establishment, Q_t^i, and forest growth conditions which differ among regions. A simplification is made by assuming a constant growth rate during time for each region, and we describe the improvement in quality during time of an established habitat in $t=0$ by an exponential function according to

$$Q_t^i = (1-e^{-\alpha^i t})(Q^{iMax} - Q_0^i)(P_0^i) + Q_0^i(P_0^i) \qquad (1)$$

where α^i is the growth rate in habitat quality in region i. As $t \to \infty$ the habitat quality of the conservation area P_0^i reaches its maximum level of Q^{iMax}, and for $t=0$ the quality at the time of establishment, $Q_0^i(P_0^i)$, prevails. The accumulated number of quality weighted habitats in a region in period *t*, S_t^i is then determined by the conservation areas P_τ^i established all periods prior to *t*, i.e. $\tau<t$, actual quality, Q_0^i, the difference between actual and potential habitat quality, $(Q^{iMax} - Q_0^i)$, and the growth rate in habitat quality, α^i.

As shown in Section 3, all parameters in (1), i.e. growth rate, and actual and potential habitat quality are calculated on the basis of data on forest structure with respect to tree age classes of deciduous forests, dead wood volume, and forest productivity. The underlying uncertainty in these variables is here captured by the region-specific estimated growth rates in habitat quality with mean μ^i and variance σ^i. The accumulated habitat quality in region *i* in a certain time, S_t^i, is then written as,

$$S_t^i = \sum_{\tau=0} (1 - e^{-\mu^i(t-\tau)})(Q^{iMax} - Q_\tau^i)(P_\tau^i) + Q_\tau^i(P_\tau^i) + \sigma_t^i \, dz \qquad (2)$$

where $\sigma_t^i = Var(S_t^i)$ and *dz* is a random parameter. In order to find the variance in (2) we carry out a Taylor expansion around the mean growth rate, μ^i, which gives

$$\begin{aligned} \sigma_t^i &= \sum_\tau Var(e^{-\alpha^i(t-\tau)} M_\tau^i) \qquad (3) \\ &= \sum_\tau Var(-e^{\mu^i(t-\tau)}(M_\tau^i - (t-\tau)e^{-\mu^i(t-\tau)} M_\tau^i(\alpha^i - \mu^i)) \\ &= \sum_\tau (-t+\tau)^2 e^{-2\mu^i(t-\tau)} (M_\tau^i)^2 \sigma^i \end{aligned}$$

where $M_\tau^i = (Q^{iMax} - Q_\tau^i)(P_\tau^i)$ and $\sigma^i = Var(\alpha^i)$. For a given conservation area P_τ^i, established at time τ, the variance in habitat quality is thus increasing for certain levels of t-τ and then approaches zero for large enough t-τ when the habitat quality reaches it maximum level. We assume independence among sites and the total variance is then the sum of variances in each region.

For each region, there exists a cost function for habitat creation, $C^i(P^i)$, which is increasing and convex in P^i. The costs consist of opportunity cost of land and management costs associated with the creation and maintenance of sufficient amount of old aged deciduous forests and dead wood. The opportunity cost is, in turn, determined by forest output and input prices which are stochastic, and payment per unit of land is therefore assumed to be uncertain. A simplification is made by assuming that the total variance in costs at a given time, σ_t^C, is the sum of cost variance in all regions, which is written as

$$\sigma_t^C = Var(\sum_i C_t^i) \tag{4}$$

Since risk in payments implies a social cost for a risk averse society, it is introduced as such in the objective function according to the Markowitz theory (e.g. Luenberger, 1998). Furthermore, the planners are assumed to be aware about the uncertainty in habitat quality growth and are concerned about the precision in reaching the habitat target. A probabilistic constraint is therefore introduced where the planner makes two choices: a minimum number of habitats to be achieved in a given time period and a minimum probability for the achievement of the target (e.g. Birge and Louveaux, 1997). We also impose constraints on areas available for the preservation of the white-backed woodpecker. The problem of minimizing total social cost under the probabilistic constraint on the habitat achievement in period T is then formulated as

$$\underset{P_t^i}{Min} \quad \sum_t \sum_i (\mu^{C^i}(P_t^i) + \theta\sigma_t^C)\delta_t \tag{5}$$

s.t. eqs. (1) - (4) and

$$\pi(S_T \geq \overline{H}) \geq \beta$$

and

$$P_t^i \leq \overline{P}_t^i \qquad for \;\; t = 1,..,T \;\; and \;\; i = 1,.,k$$

where μ^{C^i} is the mean cost, $\delta_t = \frac{1}{(1+r)^t}$ is the discount factor with r as the discount rate, θ is risk aversion, π is probability, $S_T = \sum_i S_T^i$ and S_T^i are determined according to eq. (1), β is the chosen probability for achieving the target, $\overline{P}^i$ is the maximum area available for habitat establishment in each region, and $\overline{H}$ is the target of habitat quality. The probabilistic constraint is rewritten as a deterministic equivalent (e.g. Birge and Louveaux, 1997) according to

$$\mu_T^S - \psi^{\beta} \sigma_T^{-1/2} \geq \overline{H} \qquad (6)$$

where μ_T^S is the mean impact on the habitat target, $\sigma_T = \sum_i \sigma_T^i$, and ψ^{α} is the standard for the chosen probability (the level of which depends on assumed probability distribution).

The first order conditions for a cost effective allocation of P_t^i for *t=1,..,T* and *i=1,..,k* are written as

$$\left(\frac{\partial \mu^{C^i}}{\partial P_t^i} + \theta \frac{\partial \sigma_t^C}{\partial P_t^i} \right) \delta_t - \kappa_t^i = \lambda_T \left[\sum_{\tau} (1 - e^{-\mu^i (T-t)}) - \frac{\omega^{\beta}}{2} (T-t) e^{-\mu^i (T-t)} \sigma^{1/2} \right] \frac{\partial Q_t^i}{\partial P_t^i} \qquad (7)$$

where $\kappa_t^i \leq 0$ are the Lagrange multipliers for the restrictions on P_t^i and $\lambda_T \geq 0$ is the Lagrange multiplier for the habitat target. The left hand side of (7) shows the discounted marginal social cost of a marginal habitat establishment, and the right hand is the marginal impact on the habitat restriction in the target year. The marginal cost at the LHS includes impacts on the mean and variability in costs. If the latter is positive, which is assumed in the numerical model presented in Section 3, uncertainty in costs increases total costs for achieving the target and favour establishments in regions with relatively low marginal impact on cost uncertainty. Uncertainty in improved habitat quality of a marginal area establishment has similar impacts as shown by the negative sign of the variability term within brackets at the right hand side of (7).

With respect to timing of conservation areas, the discount factor at the LHS favours late establishment of habitats since that, *ceteris paribus*, reduces total cost as measured in present terms. The impact as written on the RHS of eg. (7) consists of the mean effect and the impact on the standard deviation, the first and second expressions within the bracket, respectively. Both expressions are positive: quality and uncertainty are increasing in *(T-t)*, which favour early establishment. On the other hand, the discount rate acts in the other direction, future social costs of habitat establishments are reduced as compared with early outlays.

3. Data retrieval

The model structure presented in Section 2 reveals data needs on three types of parameters: initial and maximum quality of established habitats, Q_t^i and Q^{iMax}, quality growth, α^i, and cost functions, $C_t^i(P_t^i)$. In addition, uncertainty quantification is necessary for quality growth and costs. In the following, assessment of these data is described. In the following we give a brief presentation of the data retrieval. Unless otherwise stated, all data are described in more detail in Gren et al. (2010).

3.1. Habitat quality and growth

We assume that, in the absence of establishment of areas for white-backed woodpecker conservation, the forest structure will remain unchanged as it will be continued to be managed solely for commercial forestry purposes, which inhibits establishments of habitats with sufficient quality. Following Mild and Stighäll (2005) we define quality of white-backed woodpecker habitat in terms of hectares of mature deciduous forest and density of deadwood. When conservation areas are established, the deciduous component is allowed to age naturally and the structure becomes more mature (and hence more likely to provide white-backed woodpecker habitat). Under business as usual, there is generally too little deciduous forests of sufficient age due to the market demand for outputs from coniferous forest. The establishment of a conservation area in a given time period will then provide high quality habitat only after some period of time, the length of which varies between counties due to initial forest structure, and differences in environmental conditions and forestry practices.

An important point of departure regarding parameterization is the precise requirements of good habitat quality obtained from Mild and Stighäll (2005): 100 ha of old deciduous forest within an approximation of 500-ha large area and 20 m^3/ha deciduous deadwood for a single breeding habitat of highest quality. Therefore, to calculate habitat quality at establishment, and its subsequent growth, deciduous age class models were constructed for each county. Data on timber volumes by age class and dead wood, for spruce, deciduous, and other tree species – were available for each county, but no age-specific area coverage data were available. It was therefore assumed that the proportions of timber volume of each age-class and species combination reflected its coverage in hectares.

To calculate the initial amount of habitat quality per unit forest area within each of the twenty counties in Sweden we used estimates of forest variables produced by the Swedish University of Agricultural Sciences (Reese et al., 2003). These data were produced by combining remote sensing information from Landsat 7 ETM satellite imagery (from 1999 and 2000) with field data from a separate set of Global Positioning System (GPS)- located plots from the National Forest Inventory (NFI) using the k-nearest neighbour (kNN) method. In this method the kNN algorithm assigns to each unknown pixel the field attributes of the most similar reference pixel(s) for which field data are available (Reese et al., 2003). The kNN database used in this study consisted of a series of raster-based layers with information on forest age, tree height and estimated volume of tree species with a spatial resolution of 25 m and covered all productive forest land in Sweden. These data served to calculate variables relevant for the white-backed woodpecker habitat model. The resultant data comprised timber volumes for three species types (deciduous, spruce and others) and three age-classes (0-35, 36-70, and >70 years: “young”, “medium”, and “old” respectively). We used the known historic maximum range of white-backed woodpecker in Sweden (19th century) as a template in our analysis, which is shown in Figure A1 in the appendix (Aulén, 1988). We applied weights of 0.0, 0.25 and 1.0 to the 0-35, 36-70 and >70 year-old deciduous forest areas to calculate the age-weighted coverage of deciduous forest, dividing by 500 (hectares) to get a preliminary gross estimate of habitat. We adjusted this estimate by factors reflecting both the deadwood density and relative forest productivity of each county, to arrive at a final estimate of current white-backed woodpecker habitats in each county i (Q_0^i in eq. (1) in Section 2). Full details of our habitat calculations are given in Gren et al. 2010..

It is assumed that the relative quality of habitats among counties is determined by their actual number of habitats per unit of deciduous forest as shown in Table A1 in the appendix. Skåne (ska) then gives the largest number of habitats per thousand ha, 0.058, and Värmland (vrm) the lowest, 0.009. The initial habitat quality in Skåne (ska) is thus six times higher than that in Värmland (vrm) with respect to deciduous age-class structure, volume of dead wood, and productivity. The relative habitat quality with the Skåne quality as common denominator then varies among all counties as shown in Figure 1.

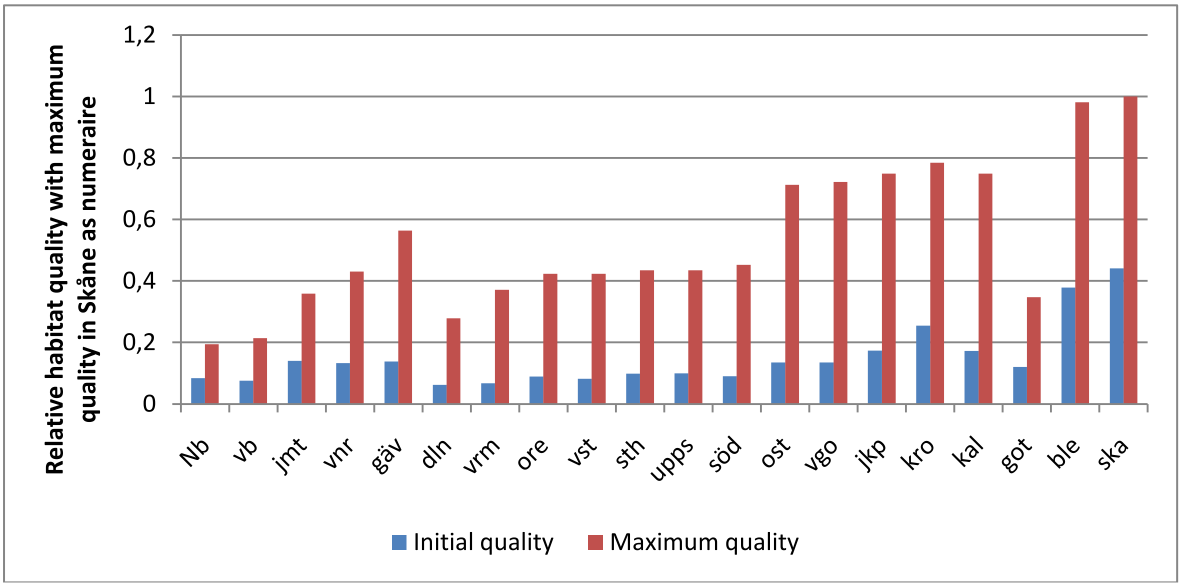


Figure 1: Relative initial and maximum habitat quality measured as number of calculated WBW habitats per unit area deciduous forests per county in relation to Skåne county (Ska).
Source: Table A1 in Appendix

The columns in Figure 1 show the initial and maximum quality in all regions in relation to the maximum quality in the most southern county Skåne (ska). The column for initial quality in this region shows that it can increase by approximately 100 per cent. When comparing initial quality among regions we find, in general, that counties located in south of Sweden show a higher quality per unit of deciduous forests with Blekinge (ble) and Skåne (ska) as outstanding. The pattern is similar for maximum quality. However, when comparing differences in initial and maximum quality we find that the growth potentials are highest in in Värmland (vrm), Västmanland (vst), Södermanland (söd), Östergötland (ost), and Västra Götaland (vgo) where initial quality can show more than a five fold growth.

Transition of forest structure in established conservation areas, i.e α^i, was calculated using an age-structured (matrix) model, with four classes: young, medium and old (as above), and a deadwood class. We parameterised the transition rates by assuming an even age-distribution within age classes, that deadwood persists for 10 years, and using natural mortality data from Ozolincius et al. (2005). Full details are provided in Appendix B in Gren et al. 2010. The behaviour of the projection model indicates that, in the conserved sites in each county i, habitat quality improves over time up to some standardized maximum level, $Q^{i,Max}$. This process is approximated by the curve described by eq. (1) in Section 2. We fitted the projected forest data for each county to eq. (1), by setting Q_0^i and Q^{Max} and using the "LSQCURVEFIT" optimization function in MATLAB (release 2009b, version 7.9.0; The Mathworks Inc.) to minimize the sum of squared errors in estimating α^i. The parameters for each county's habitat accumulation curve are given in Table A2 in the appendix

Figure 2 shows a variation in annual growth among counties that ranges between 0.0269 (Västernorrlands, vnr) and 0.0512 (Skåne, ska). This is a significant difference when considering the accumulated impacts during a 60 year period. Establishment of habitat in Skåne (ska) will reach a given habitat quality more rapidly than in Västernorrland (vnr) which implies a lower cost.

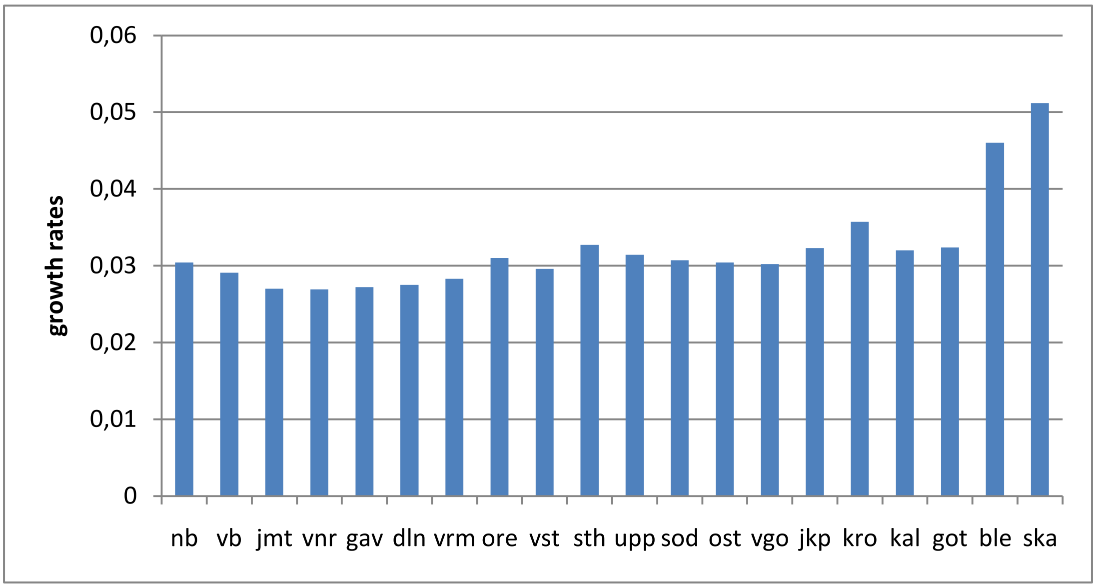


Figure 2: Mean annual growth in habitat quality for different counties
Source: Table A2 in appendix A

We incorporate uncertainty in habitat projection by assuming that the true values of the non-standardized Q_0^i and Q^{iMax} lay within ±50% of our estimates from the data). While this 50% error term is somewhat arbitrary, we feel that its magnitude reflects the many broad assumptions necessary to convert current woodland data into present and future white-backed woodpecker habitat units. These deviations in initial and maximum quality generate a range of growth rates for each county, see Table A2 in appendix. We quantify uncertainty in growth rate for each county as the mean divided by the range, see Figure 3.

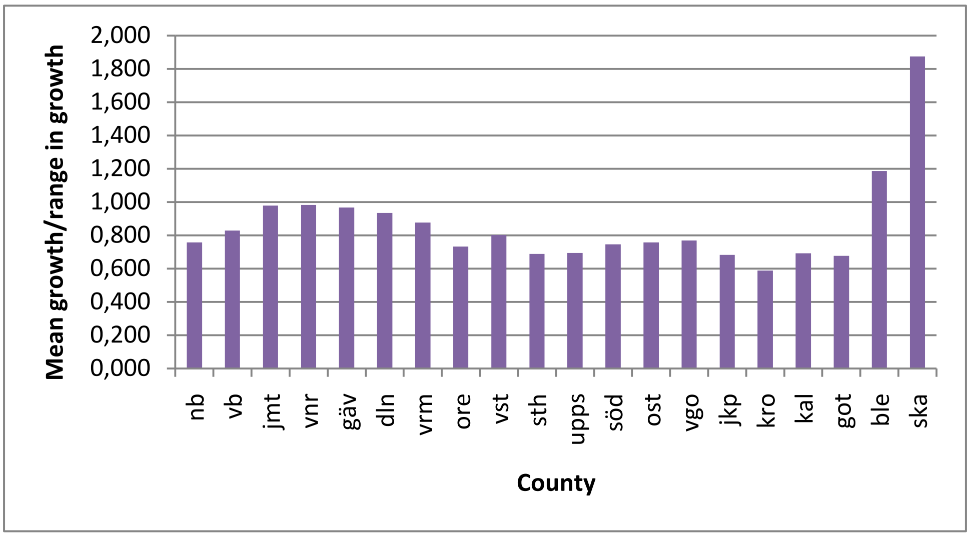


Figure 3: Uncertainty in provision of habitats with sufficient quality among counties
Source: Table A2 in the appendix

The difference in uncertainty, quantified as the range divided by the mean growth, among counties is higher than that in average growth rate. Skåne (ska) and Blekinge (ble) counties constitute the most uncertain habitat investment regions, while other southern regions, Kronoberg (kro) and Gotland (got), turn out to be the most safe investment regions.

3.2 **Cost functions**

As described in Section 3.1 suitable habitats are obtained by management of forest land providing dead wood, deciduous forests in specific age classes etc. The cost components from these activities include management cost and opportunity cost of the forest land. Since the late 1990s voluntary agreements between forest owners and the Swedish Forestry Board have been reached where the owners receive compensation payments for appropriate management for habitat provision. Gren and Carlsson (2010) carried out econometric estimates of cost functions for habitat provision based on these actual compensation payments and areas of habitats on a panel data set covering all counties during the period 1998-2009. Total compensation payments in each county constituted the dependent variable and explanatory variables were derived under the assumption of a typical forest owner's maximisation of current and future streams of net utility from the land use under business as usual. It was then assumed that an owner does not accept a compensation payment unless this covers the cost – management and opportunity cost – of compliance with the agreement. This cost is, in turn, determined by the land owners' utility function encompassing net benefits from commercial use of the land, but also environmental preferences and eventually other income opportunities. Considering these factors Gren and Carlsson (2010) introduced output prices of forest products, wage rate, interest rate, environmental attitudes, and regional economic development as explanatory variables. A random effect model was applied to regional clusters of counties reflecting differences in forest growth conditions. The estimated coefficients together with areas of conservation agreements are presented in Table A3 in the Appendix A. Figure 4 displays marginal cost of area provisions in the different counties.

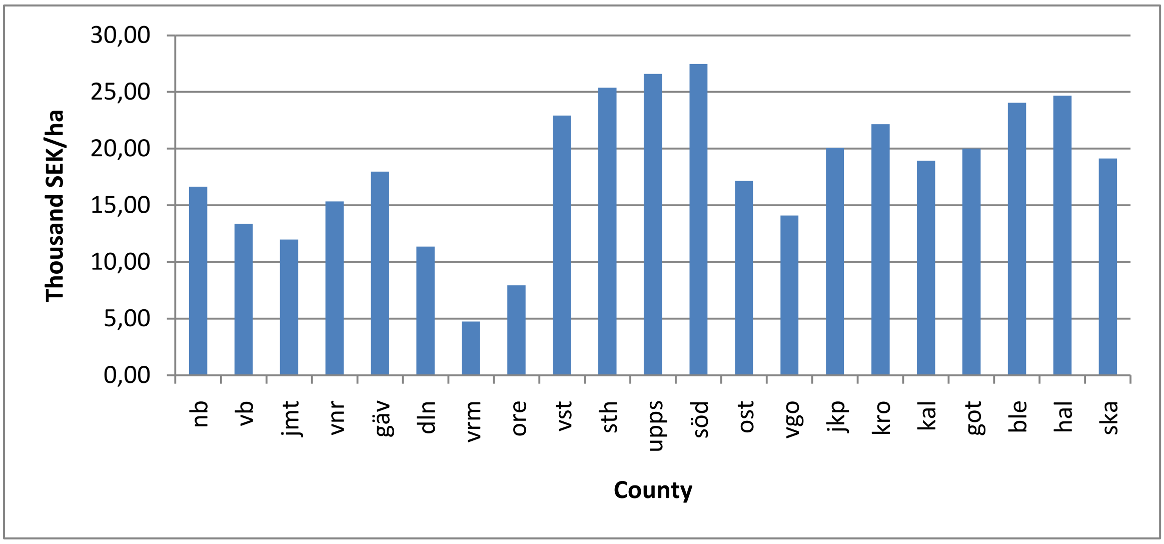


Figure 4 : Calculated marginal provision cost in different counties, 1000 SEK/ha
Source: Table A3 in the appendix

Figure 4 shows considerable differences in marginal provision cost ranging from approximately 5000 SEK (1 Euro = 8.97 SEK January 30, 2011) in Värmland (vrm) county to 26 000 SEK in Södermanland (söd). In general, the marginal provision costs are low in the north of Sweden as compared with the densely populated regions in the south of Sweden.

However, recall from the theoretical Section 2 that the unit payments are uncertain, which can reflect fluctuations in market prices of forest land, expected incomes etc. We quantify this uncertainty as the coefficient of variation in actual payments in each county during the period 1998-2009, which are shown in Figure 5.

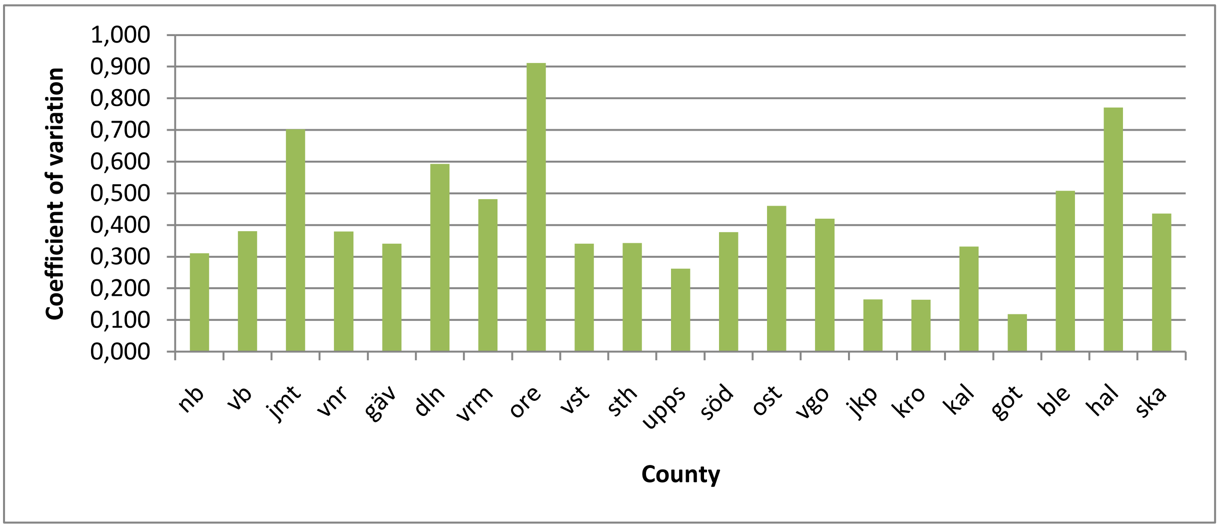


Figure 5: Coefficient of variation in payments per ha for different counties

Source: Table A3 in appendix

Interestingly, the pattern of quantified uncertainty differs from that of estimated marginal provision cost: coefficients of variation in the northern regions are now at the same level or even larger than those in the south. The highest level is found for a county (Örebro, öre) with relatively low marginal provision cost and counties with relatively high marginal provision cost reveal the lowest coefficients of variation (Kronoberg, kro, Gotland, got, and Jönköping, jkp). Thus, when accounting for uncertainty in provision costs a risk averse biodiversity manager may choose seemingly expensive site locations.

In addition to cost functions and uncertainty quantification we need to determine maximum areas of habitat provision in each county, discount rate, level of risk aversion and desired probability of achieving the target. Since the estimated cost functions are defined for a maximum area for each county, these areas are used as a restriction on habitat provision for each year. We apply a real discount rate of 0.03 in the reference case since this is close to the rate of return on risk free governmental bonds during the last 20 years. Following Alvarez et al (2007) we apply a 0.001 as the reference value of risk aversion. We finally assign the chosen probability of achieving the target in the reference case to 0.95 and assume a normal probability distribution.

4. Results: optimal spatial and dynamic site selections

Recall from the introductory section that the Swedish Environmental Protection Agency aims at achieving habitats for 200 pairs of WBW in year 2070. In principle, the choice of optimal allocation of sites among counties for achieving this target would be relatively easy if high habitat growth, low marginal provision cost, and low uncertainty in growth and provision cost are positively correlated among the counties. However, as shown in Section 3 this is not the case, and we therefore solve the optimal allocation of sites by means of GAMS software (Brooke et al., 1998).

Since the main purpose of this paper is to investigate the role of uncertainty in growth of number of habitats with sufficient quality and in provision costs we present results under deterministic and stochastic conditions of provision payments and of habitats. Figure 6 displays total costs for the entire period under four different uncertainty combinations.

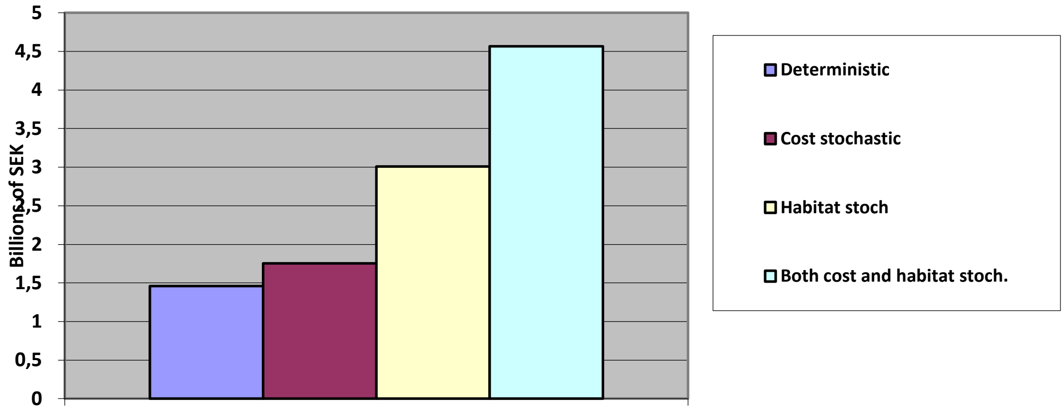


Figure 6: Total minimum social costs under deterministic and different stochastic combinations, billions of SEK

The results indicate considerable differences between total social costs depending on assumed uncertainty. The out of pocket costs under combined uncertainty are lower and correspond to 1.5 billion SEK (1 Euro = 8.97 SEK, January 30, 2011) under only cost uncertainty and to 3.8 billion SEK when both types of uncertainties act. The high cost shown in Figure 6 under combined uncertainty is then explained by the need to establish larger amount of habitats due to the probabilistic constraint and to risk aversion in costs. We note also from Figure 6 the impact of habitat uncertainty, which results in an 'excess' safety investment in habitats corresponding to approximately 40 percent of the required 200 habitats. This excess investment explains the increase in total cost from 1.46 to 3.07 billions of SEK when moving from the deterministic to the habitat uncertainty case.

However, even though the total costs differ under the four different stochastic combinations, the patterns of habitat establishment over time are more similar, see Figure 7.

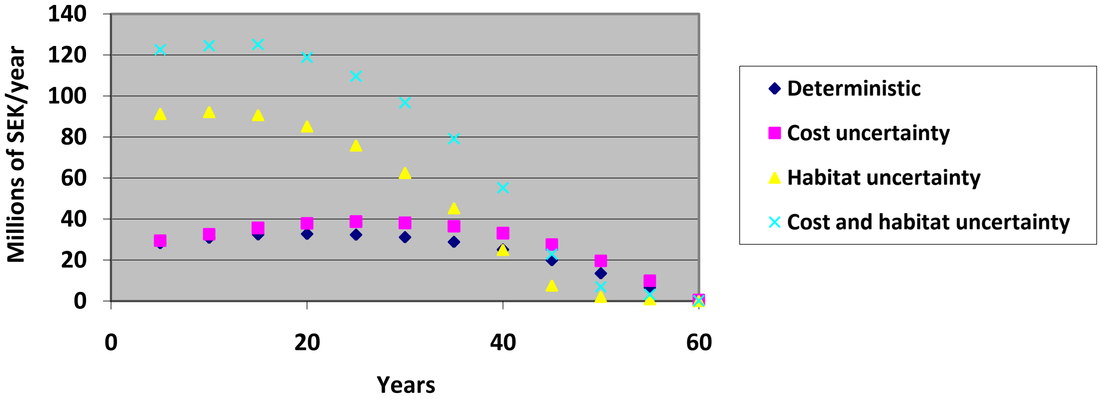


Figure 7: Optimal paths of annual social costs in present terms under different conditions of cost and habitat uncertainty

As expected, the levels of annual social costs are higher under conditions of habitat uncertainty as compared with the deterministic case and that with only cost uncertainty. The difference in pattern of payments during time is somewhat less obvious. There is a longer delay in social costs without habitat uncertainty. This is because of the discount rate which acts in favour of delayed costs. The earlier acquisition of land and hence costs under habitat uncertainty is due to the need for 'safety' investment in extra habitats, which is shown in Figure 8.

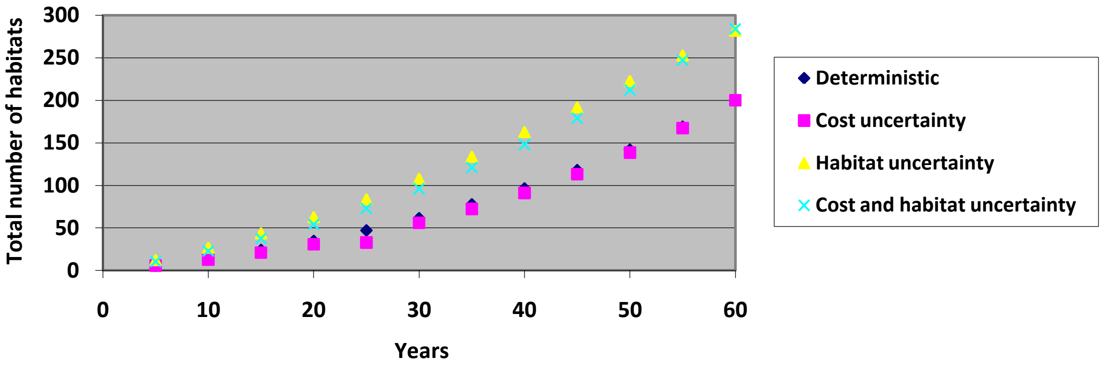


Figure 8: Time paths of socially cost effective habitat provision under different stochastic assumptions.

Figure 8 shows the results of extra habitat establishment corresponding to approximately 1/3 of the required 200 habitats, i.e. in order to achieve 200 habitats in 2070 with a probability of 0.95 a minimum of 280 habitats have to be established. The shape of the habitat growth functions does not allow for a decline in quality, but only a decline in incremental quality during time. The conservation areas shown in Figure 9 then generate decreasing marginal improvements in habitat amount for additional 80-100 years after the target year.

It is also interesting to note that the relative allocation of habitats among counties shows a similar pattern under the deterministic and stochastic cases, see Figure 9.

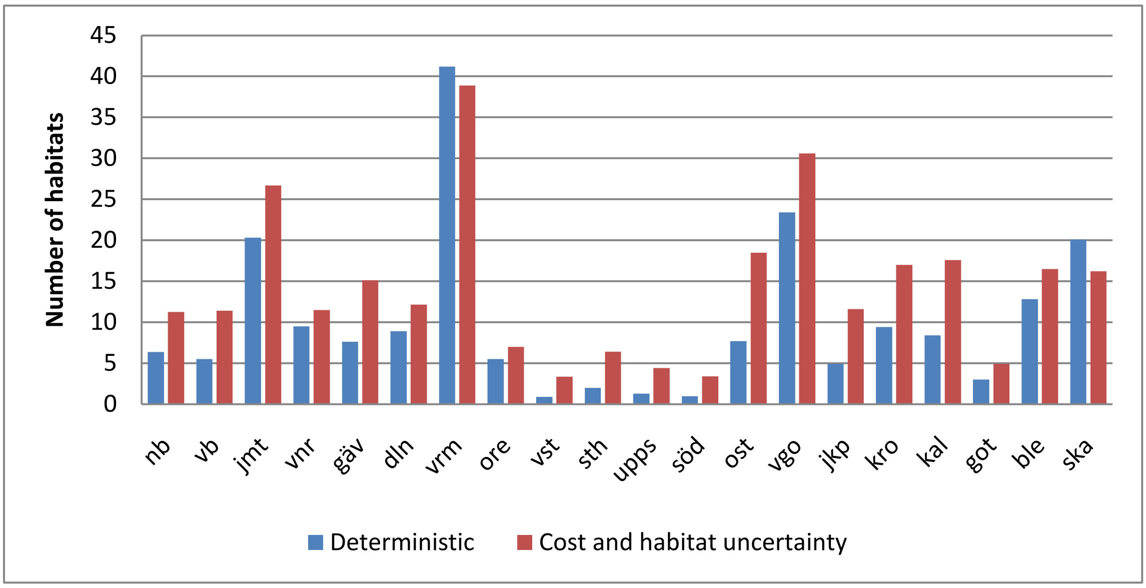


Figure 9: Allocation of habitat establishments among counties under conditions of deterministic and combined uncertainty

The largest amount of habitats is established in the Värmland (vrm) county under both stochastic combinations, and the smallest number in Västmanland (vst) and Södermanland (söd). We can also note the increase in habitat provision under stochastic conditions in counties with relatively low habitat and cost uncertainty: Jämtland (jmt), Östergötland (ost), Kronoberg (kro), and Kalmar (kal).

Recall from Section 3 our assumptions with respect to choice of discount rate, risk aversion against variability in social costs, and the chosen probability of achieving the target. Changes in these parameter values will change total costs. In Figure 10 we show impacts on costs from changes in these assumptions.

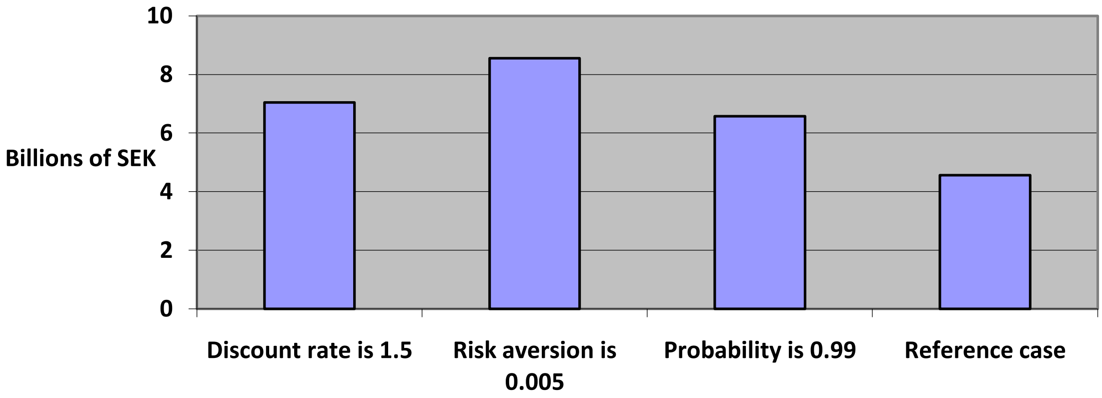


Figure 10: Impacts of total costs from a decrease in discount rate, increase in risk aversion, and in the chosen probability of achieving the target when both cost and habitat improvement are uncertain

The change in the discount rate is reduced by one half compared with the references case, the risk aversion is increased five-fold, and the probability of achieving the target increases from 0.95 to 0.99. Total acquisition cost increases from all these parameter changes. Similarly, the cost decreases for an increase in discount rate, and for decreases in risk aversion and probabilities of achieving the target.

5. Policy analysis

In general, policy makers have to take the functioning of nature and human decisions as given, at least in the short run. The main policy parameters are then the choices of target formulation – amount and timing of habitats – and choice of policy instrument for the implementation of cost effective solutions. Economic analyses of both these types of choice parameters have occupied a large research field in environmental economics for decades. It is well known that costs increase in the stringency of the target and decrease in the time delay of the implementation of the target. Similarly, there is a considerable literature on the efficient design of payments for biodiversity preservation which accounts for heterogeneous habitat sites, uncertainty in reaching the targets, and asymmetric information (e.g. xx). A major lesson from this literature is that it is most often not possible to implement the cost effective allocation of habitat sites due to high monitoring and enforcement costs. The reason is the need to design and supervise policies, mainly compensation payments, for each type of site. These transaction costs can be significant and correspond to the same amount per unit area as the provision costs in terms of management and opportunity cost (e.g. Vatn, 2010).

We will in this chapter present results related to the first type of policy parameter, i.e. calculation of costs for alternative choices of number of habitats and the timing of target achievement. With respect to the second type of policy issue, we will investigate of the optimal policy design in different time periods and among counties and compare this scheme with actual payments during the period 1998-2009.

5.1 Stringency and timing of habitat target

With respect to the stringency of target, i.e. number of habitats in 2070, results displayed in Figure 11 reveal higher increase in total costs under habitat uncertainty, which is due to the need of a larger number of habitats to ensure achievement of 200 successful establishments of WBW with a probability of 0.95.

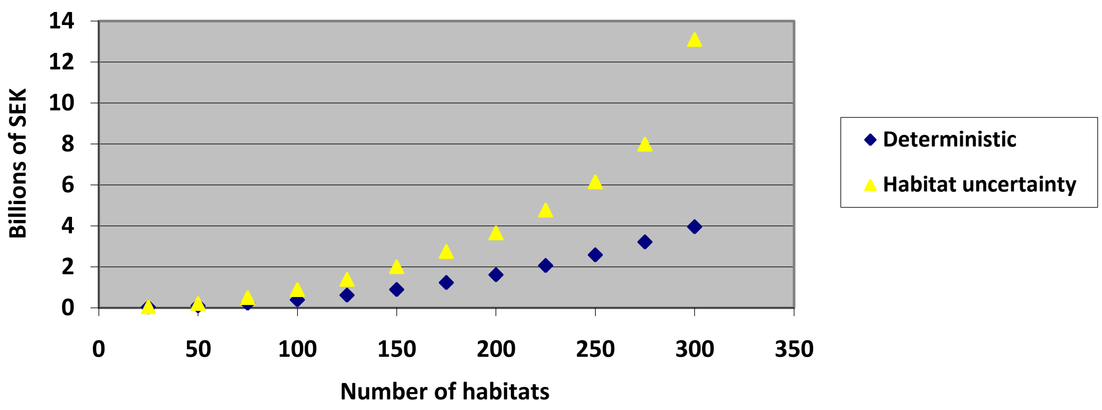


Figure 11: Minimum costs for different number of habitats in 2070 under deterministic and habitat uncertainty conditions (with ψ^{a} =0.95).

The figure also shows that a change (increase or decrease by 25 habitats from the main target of 200) may increase/decrease total costs by approximately 35/24 percent under the deterministic case and with 30/25 percent under habitat uncertainty. Although the relative change in costs is lower under habitat uncertainty conditions, a decrease in the target by 25 habitats implies cost saving of approximately 6 million SEK/year in present terms during a 60 year period.

Figure 12 shows that cost savings can also be made by delaying the time of target achievement, and vice versa.

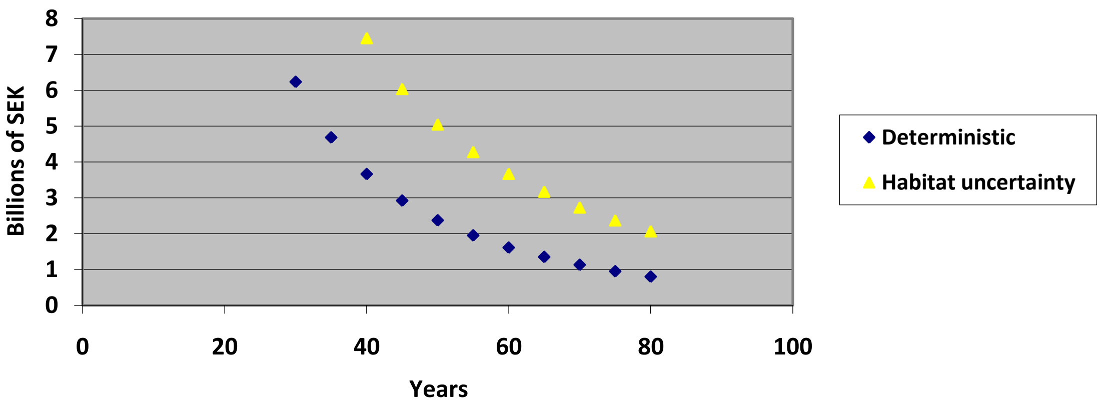


Figure 12: Costs for different timings of achievement of 200 habitats under deterministic and uncertainty conditions (ψ^a =0.95)

The results presented in The results presented in Figure 12 show that if the time of the target achievement could be delayed by 10 years, society would obtain cost savings corresponding to approximately 30 per cent under both stochastic cases. The cost increase from a 10-year earlier achievement of the target amounts to approximately 40 per cent of the reference cost. Thus, a combination of both earlier and more stringent target can increase the cost considerably.

5.2 Policy design

The Lagrange multiplier λ_T presented in the theoretical chapter 2 constitutes the point of departure for cost effective design of compensation payments. As shown in Chapter 2 this design is characterised by the impact on the target and on the marginal provision cost.

The higher the impact, *ceteris paribus*, the higher is the compensation payment. The impacts on the target from each county are determined by the quality parameter and the growth rates in habitat quality and, under stochastic conditions, uncertainty in growth. In a cost effective solution the optimal payments are given by the level of the Lagrange multiplier and the unit compensation payments correspond to the impact on the target, times the Lagrange multiplier. Figure 13 displays that the Lagrange multiplier increases rapidly at habitat targets exceeding 225 under habitat uncertainty.

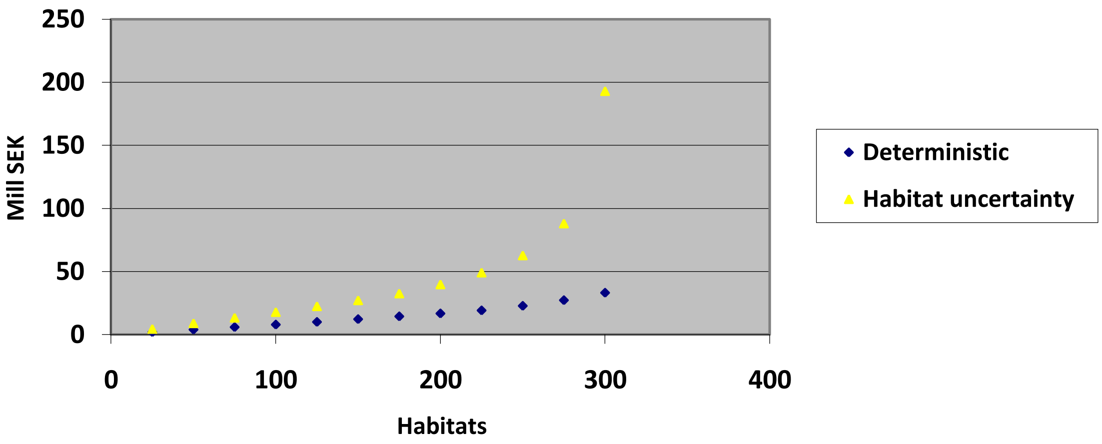


Figure 13: Lagrange multiplier, or marginal cost for achieving different numbers of habitats in 2070

At the reference target of 200 habitats the Lagrange multiplier is approximately 17 and 39 millions of SEK under the deterministic and stochastic cases. That is, the establishment of one additional habitat would increase total costs by 17 or 39 millions of SEK. The optimal compensation payments increase over time due to the higher impact on the target. Changes in target year have also considerable impact on the Lagrange multipliers, which can be seen from Figure 14.

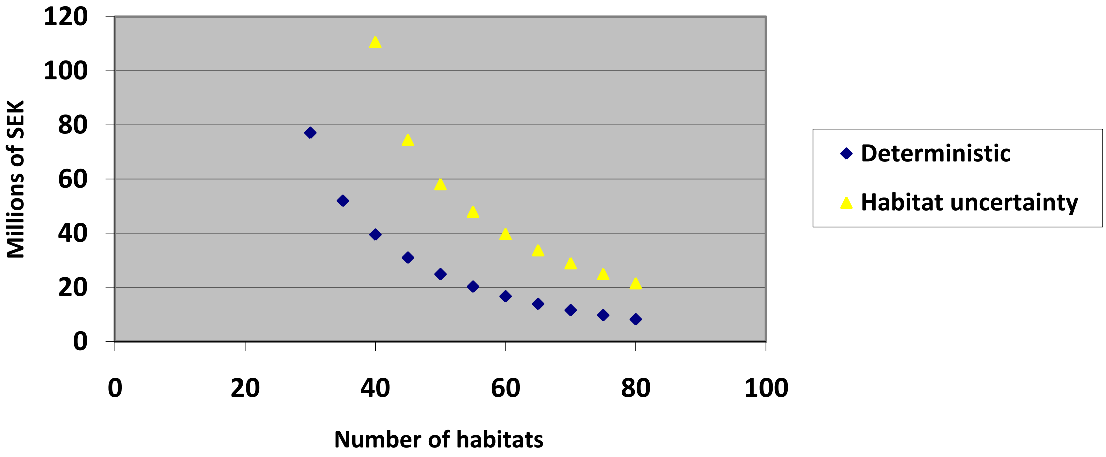


Figure 14: Lagrange multipliers, increase in total cost from increasing the habitat requirement by one unit, at different target years.

For relatively early target year, 30 years from now, the Lagrange multiplier increases by approximately five times for the deterministic case and three times under habitat uncertainty as compared with the reference cases. On the other hand, a two-decade delay in target year reduces the cost at the margin by approximately one half compared with the reference year.

When comparing actual payments per ha during the period 1998-2009 with the cost effective payments under deterministic conditions it is interesting to note that they coincide for several counties; Jämtland, Värmland, Örebro, Jönköping, and Kalmar. However, the mean payments under uncertainty conditions are higher than actual mean payments for these counties and also for most others. For four counties – Norrbotten, Västmanland, Stockholm, and Uppsala – the actual mean payments are quite close to the cost effective payments under conditions of cost and habitat uncertainty. We can thus conclude that the actual payments seem to be based on economic rationality for some counties, but the underlying decision rules (with or without consideration of uncertainty) might differ. For some counties – Västergötland and Södermanland – actual mean payments are higher or equal to payments under any of the decision rules.

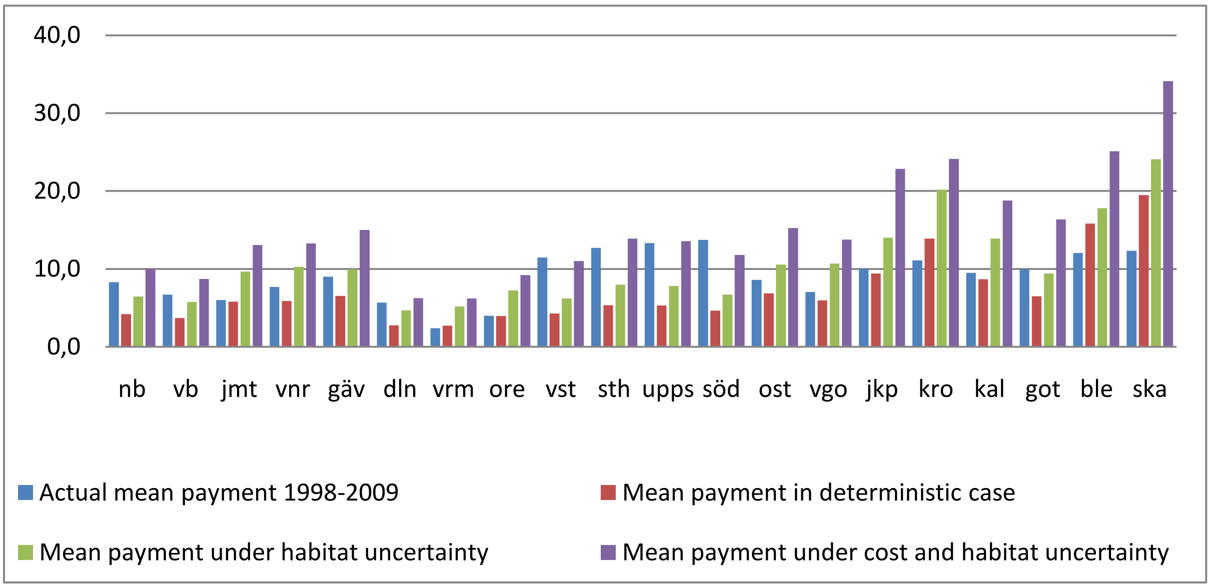


Figure 15: Actual average payment during 1998-2009, and cost effective mean payments during 2010-2070 under deterministic and uncertainty conditions, thousand SEK/ha.

6. Conclusions

The main purpose of this paper has been to calculated optimal location of conservation sites selection for an umbrella species – white-backed woodpecker – in Sweden. A specific feature of our paper is the inclusion of heterogeneous conditions in different regions of Sweden with respect to habitat quality, which is described by areas covered by old deciduous forests and dead wood. Another noteworthy contribution is the recognition and quantification of uncertainty both in development of habitat quality over time of an established site and the stochastic costs for land owners because of fluctuations in land market prices and labour costs. A combination of economic and ecological modelling is applied where a dynamic stochastic model is constructed for determining the optimal path of number of habitat establishment in different regions during time. An ecological model is used to parameterize average habitat growth in different regions and to quantify uncertainty as the coefficient of variation in growth rate.

The description of the regions with respect to habitat quality and provision cost shows large variation. However, the performance of the regions differs depending on quality, growth in habitat quality, costs, and uncertainty. For example, regions located in south of Sweden show relatively high quality, but also high acquisition cost and large variation in development of quality during time. Therefore, since there is no region which is best with respect to all parameters – i.e. high quality, high growth in quality during time, low costs, and low uncertainty – a stochastic and dynamic model is constructed which solves for the optimal site selection in space and time. The main results are:

- The optimal average discounted annual social cost varies between 24 and 76 millions of SEK depending on assumption of uncertainty. The costs are most sensitive to uncertainty in development of habitat quality.

- The pattern of social costs during time differs depending on the included uncertainty: habitat uncertainty requires earlier establishments due to the need for 'safety'investment.

- Optimal average annual payment per hectare of established habitat varies between approximately SEK 6000/ha and 34000/ha among regions when uncertainty in both costs and habitat quality is considered.

- When comparing optimal payment per hectare conservation with actual payments, there is a considerable difference among counties; while actual payments for some counties are quite close to optimal payments under any of the uncertainty conditions, they can deviate largely for some other counties.

All the listed results are obtained in the reference case with assumptions on risk aversion, discount rate, required probability of achieving the targets, choice of time for achieving the target, and the targeted number of habitats. Results from sensitivity analyses show that total social costs are highly affected by changes in these parameters. Nevertheless, the results are robust with respect to the role of heterogeneous regions and impacts of uncertainty in costs and development of habitat quality.

Although the numerical model extends earlier empirical studies by including uncertainty in both development of habitat quality and costs it does not address the role of connectivity, or its inverse, isolation, of habitats. This is widely used in spatial ecology and has been tested as a fundamental cause of species dispersal in several studies (see meta analyses in Molianen and Nieminen, 2002). The consideration of connectivity will by all likelihood impact the modelling and associated results in this paper by affecting our measurement of habitat quality, which would need to consider the linkages between patches and habitats. Since the borders of these agglomerations of habitats are likely not to follow those set by the jurisdictional units, cooperation among counties might be necessary for providing optimal location and timing of habitats.

Appendix A: Tables and figures

Table A1. Summary data, results of projection of current woodland state, and habitat model parameters for each county for the calculations of habitat quality in growth

County name, code	Total deciduous cover (ha)	Estimated current WBW habitats	Estimated maximum future habitats based on current habitats	Initial habitat per million hectares established (q_0)	Max habitat per million hectares forest established (q_{max})
Norrbottens, nb	536303	5,958	13,707	11,1095	25,5576
Vaesterbottens, vb	559239	5,572	15,770	9,9630	28,1981
Jaemtlands, jmt	261079	4,836	12,345	18,5234	47,2858
Vaesternorrlands, vnr	420932	7,342	23,883	17,4413	56,7391
Gaevleborgs, gav	290960	5,294	21,618	18,1948	74,2983
Dalarnas, dln	300567	2,477	11,016	8,2420	36,6501
Vaermlands, vrm	269100	2,378	13,150	8,8357	48,8682
Oerebro, ore	119232	1,407	6,647	11,8001	55,7518
Vaestmanlands, vst	81126	0,875	4,522	10,7834	55,7455
Stockholms, sth	86166	1,119	4,936	12,9883	57,2849
Uppsala, upp	106088	1,390	6,077	13,1019	57,2808
Soedermanlands, sod	68964	0,821	4,108	11,9044	59,5690
Oestergoetlands, ost	128613	2,279	12,089	17,7180	93,9914
Vaestergoetalands, vgo	105995	1,881	10,087	17,7463	95,1655
Joenkoepings, jkp	76642	1,747	7,565	22,7975	98,7038
Kronobergs, kro	68440	2,296	7,079	33,5527	103,4259
Kalmar, kal	141105	3,201	13,927	22,6828	98,7027
Gotlands, got	23546	0,373	1,079	15,8417	45,8292
Blekinge, ble	35586	1,774	4,603	49,8640	129,3471
Skane, ska	56874	3,309	7,499	58,1860	131,8527

Table A2: Average, minimum, maximum, and range in calculated growth in habitat

County	*Average growth*	*Min growth*	*Max growth*	*Range*	*Coeff. of vari*[1]*.*
nb	0.0304	0.0295	0.0525	0.023	0.189
vb	0.0291	0.0277	0.0518	0.024	0.207
jmt	0.0270	0.0243	0.0507	0.026	0.244
vnr	0.0269	0.0243	0.0507	0.026	0.245
gäv	0.0272	0.0247	0.051	0.026	0.242
dln	0.0275	0.0253	0.051	0.026	0.234
vrm	0.0283	0.0265	0.0513	0.025	0.219
ore	0.0310	0.0302	0.0529	0.023	0.183
vst	0.0296	0.0284	0.0521	0.024	0.200
sth	0.0327	0.0306	0.0531	0.023	0.172
upps	0.0314	0.0322	0.054	0.022	0.174
söd	0.0307	0.0298	0.0527	0.023	0.186
ost	0.0304	0.0295	0.0525	0.023	0.189
vgo	0.0302	0.0292	0.0524	0.023	0.192
jkp	0.0323	0.0317	0.0537	0.022	0.170
kro	0.0357	0.0353	0.0563	0.021	0.147
kal	0.0320	0.0314	0.0535	0.022	0.173
got	0.0324	0.0319	0.0538	0.022	0.169
ble	0.0460	0.0408	0.0953	0.055	0.296
ska	0.0512	0.0305	0.1265	0.096	0.469

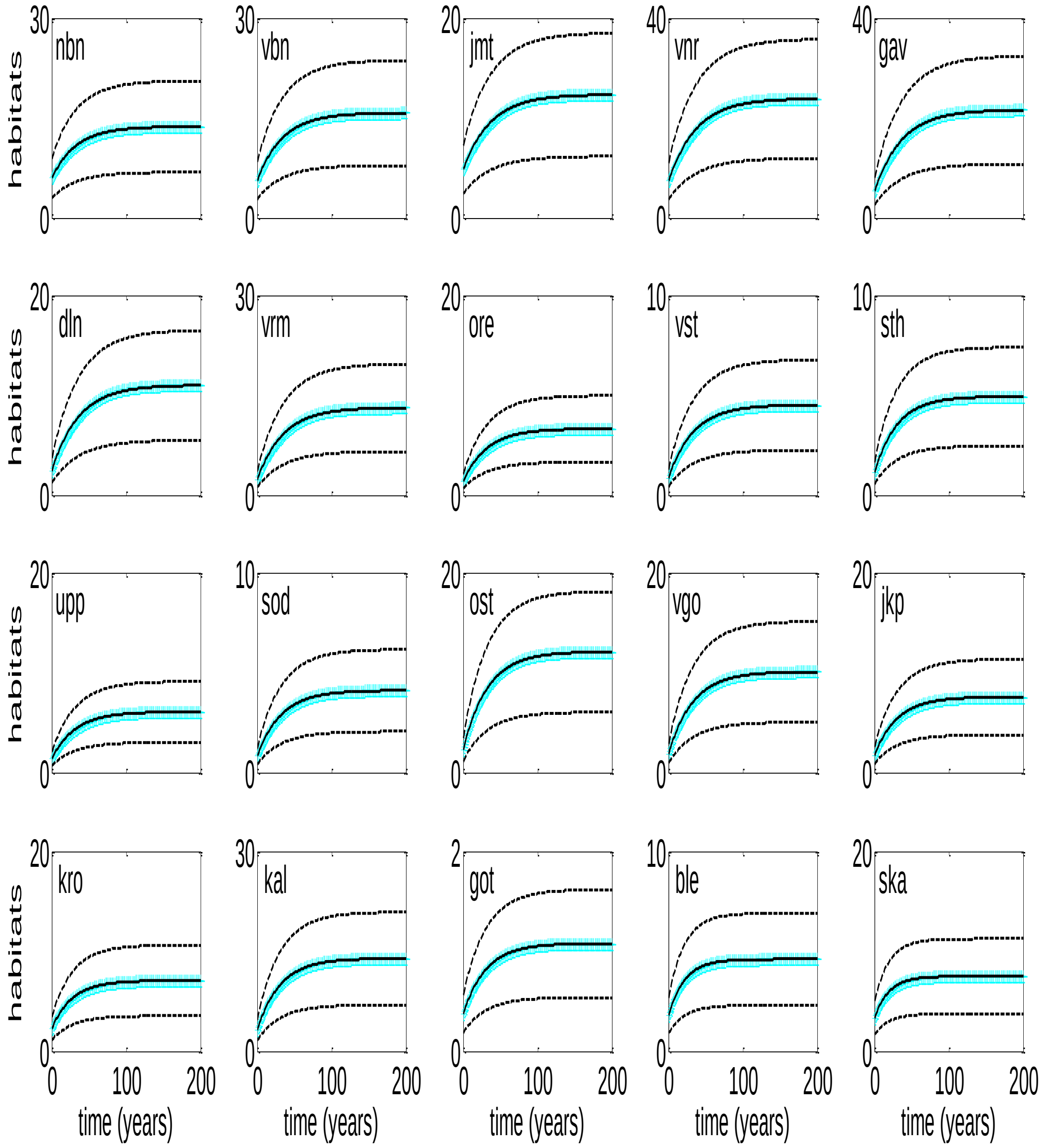
1. Assuming 95 % probability and normal distribution.

Table A3: Conservation areas, estimated coefficients in the quadratic cost function, and marginal provision cost at the mean ha values (averages during 1998-2009)

	Average number of established ha during 1998-2009	*Average payment, 1000 SEK/ha during 1998-2009*	*Estimated coefficient in quadratic cost function*	*Calculated marginal provision cost, 1000 SEK, at the mean level*	*Coefficient of variation*
nb	239	8,32	0,0348	16,64	0,311
vb	208,6	6,69	0,0321	13,37	0,381
jmt	230	5,99	0,0260	11,97	0,702
vnr	121	7,68	0,0635	15,36	0,379
gäv	116	8,99	0,0775	17,98	0,341
dln	425	5,67	0,0133	11,35	0,593
vrm	720	5,03	0,0033	10,05	0,481
ore	94,6	3,96	0,0419	7,93	0,911
vst	51,5	11,46	0,2225	22,91	0,342
sth	78,25	12,69	0,1622	25,38	0,343
upps	53,4	13,30	0,2490	26,59	0,262
söd	47,92	13,73	0,2866	27,47	0,378
ost	114,67	8,57	0,0748	17,14	0,460
vgo	290,2	7,04	0,0243	14,09	0,420
jkp	50,92	10,04	0,1971	20,07	0,165
kro	50,82	11,08	0,2180	22,16	0,164
kal	84,83	9,47	0,1117	18,94	0,332
got	62,36	10,00	0,1603	20,00	0,119
ble	46,67	12,03	0,2578	24,06	0,508
hal	37,55	12,33	0,3284	24,66	0,771
ska	37,83	9,57	0,2530	19,14	0,436

Source; Gren and Carlsson 2010

Figure A1. Modelled habitat accumulation of current deciduous woodland in twenty counties throughout Sweden, if managed for white-backed woodpecker. Blue lines show the modelled projection of actual forest data for each county; solid black lines are the best-fit curves (fitted to eqn 7); and dashed black lines indicate the range of uncertainty assumed, from 50% to 150% of initial and final habitat quality. Counties are indicated by three-letter code (see Table A1).



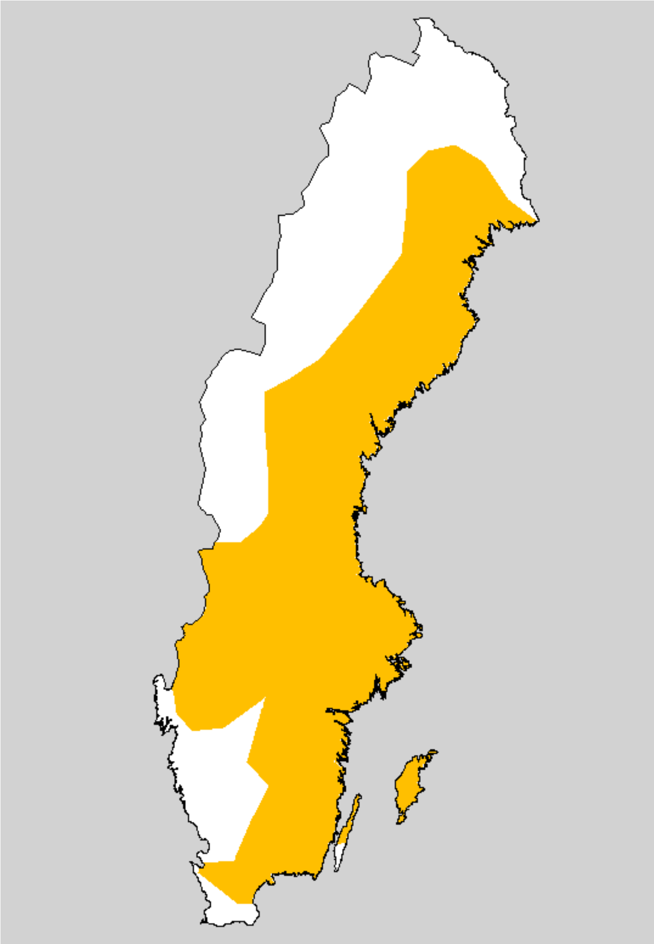


Figure A2: maximum known historic range of the white-backed woodpecker in Sweden. Based on maps in Aulén (1988) and Mild and Stighäll (2005).

Appendix B. Calculations of White-backed Woodpecker habitat accumulation

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This appendix describes the habitat calculations, as follows:

(i) the interpretation of forest data (Reese et al. 2003) to estimate current white-backed woodpecker habitat quality; and

(ii) the projection of the forest structure under conservation agreements to provide future WBW habitat.

B1: Current WBW habitat quality

We obtained forest cover data (Reese et al. 2003) for the twenty Swedish counties lying within the maximum known historic range of the white-backed woodpecker (Fig. A2). We partitioned the area (hectares) of productive forest in county *i*, by age-class (*x*), calling the partitioned areas $A_{x,i}$, with $x = Y, M, O$ representing "young" (0-35 years), "medium" (36-70 years) and "old" forest (>70 years) respectively. We also partitioned the timber volume data (Reese et al. 2003), giving $V^{(D)}{}_{x,i}$, and $V^{(T)}{}_{x,i}$, the volumes of deciduous and total forest, respectively, in each county *i* and age-class *x*.

By assuming that proportions of volume of age-classes are representative of the proportions of the coverage area of each age-class, we can estimate the extent of coverage, in hectares, of young, medium, and old deciduous forest in each county:

$$a_{x,i} = A_{x,i} \, (V^{(D)}{}_{x,i} \, / \, V^{(T)}{}_{x,i}) \qquad \text{(B1)}$$

As these age-classes roughly correspond with meeting woodpecker requirements poorly, moderately or well, we weighted the areas by 0.00, 0.25 and 1.00 respectively, to obtain the age-weighted deciduous component in county *i*, W_i :

$$W_i = \textstyle\sum_x w_x \, a_{x,i} \qquad \text{(B2)}$$

where $w_1 = 0$; $w_2 = 0.25$; $w_3 = 1.00$. The values W_i can be interpreted as the amount of "gross" habitat per county, before adjustment for any productivity or deadwood component.

To account for regional variations in climate, site and forestry practices we adjusted the gross habitat figures for relative productivity and deadwood density in the county. We obtained productivity data expressed by mean annual volume increment in years 2005-2009 (m^3 ha^{-1} yr^{-1}) from the Swedish National Forest Inventory (SNFI) (www.slu.se/en/webbtjanster-miljoanalys/forest-statistics/ - Table 16SD), which gave an estimate of forest production for each county, Fi. We scaled the gross habitat of each county, Wi, by the productivity of the county relative to that of Skåne, the most productive county, i.e. by a factor of F_i / $F_{\text{Skåne}}$. Relative productivities ranged from 0.26 in the northern county Norrbotten, to 0.98 for Blekinge (and 1.0 for Skåne) in the south.

Deadwood density data were available at a broader regional level, with only four estimates for the whole of Sweden was provided by SNFI and based on field measurements from 2005-

2009 (www.slu.se/en/webbtjanster-miljoanalys/forest-statistics/ - Table 25SD). We set deadwood density in each county, ρ_i, to the relevant regional deadwood density. As white-backed woodpeckers require 20 m^3 of deciduous deadwood per hectare (Mild & Stighäll 2005), we further scaled the gross habitat in each county by $\rho_i/20$. Adjusting the gross habitat for these variations in productivity and deadwood yielded our estimates of the current habitat in each county, $h_{0,i}$:

$$h_{0,i} = (F_i \,/\, F_{\text{Skåne}})\,(\rho_i \,/\, 20)\, W_i \tag{B3}$$

B2. WBW habitat projection

We assume that if land is unconverted it remains actively managed for forest, and that the structure remains unchanged. Note that this assumption also validates the use of the 1999-2000 forestry data (Reese et al. 2003) to calculate current habitat as above. In contrast, we assume that with establishment of conservation areas—when land is acquired for WBW management, through biotope management or conservation agreements—the deciduous component of that land matures naturally and provides more WBW habitat over time. Therefore we only project habitat improvement for that land under biotope management or conservation agreements (between which we make no distinction in this paper [!]).

We use an age-structured model to project the forest structure in conserved areas over time, calculating the initial age-structure as in B1 above. In each county, we assume that $a_{Y,t+1}$, the area of forest that is "young" (in the 0-35 year age-class) in year t+1 depends on the current area of young forest ($a_{Y,t}$) and recruitment in the space vacated by the decay of deadwood:

$$a_{Y,t+1} = s_Y\, a_{Y,t} + r_D\, a_{D,t} \tag{B4}$$

where s_Y is the retention ("survival") of young forest in that age-class, and r_D is the rate of decay of the current deadwood, covering area $a_{D,t}$. The area of medium-aged forest, $a_{M,t}$ depends on growth of young forest and retention rate s_M of the medium-aged class:

$$a_{M,t+1} = s_M\, a_{M,t} + g_Y\, a_{Y,t} \tag{B5}$$

where g_Y is the growth rate of young forest. The area of old forest, $a_{O,t}$ similarly depends on growth of medium-aged forest (at rate g_M) and retention rate s_O of the old-aged class:

$$a_{O,t+1} = s_O\, a_{O,t} + g_M\, a_{M,t} \tag{B6}$$

The forest area covered by deadwood, $a_{D,t}$ depends on the natural mortality of all age-classes (d_j, with j = Y, M, O for young, middle-aged and old forest respectively) and retention of deadwood:

$$a_{O,t+1} = (1 - r_D)a_{D,t} + d_Y\, a_{Y,t} + d_M\, a_{M,t} + d_O\, a_{O,t} \tag{B7}$$

The model can be re-written in matrix notation as

$$\begin{pmatrix} a_Y \\ a_M \\ a_O \\ a_D \end{pmatrix}_{t+1} = \begin{pmatrix} s_Y & 0 & 0 & r_D \\ g_Y & s_M & 0 & 0 \\ 0 & g_M & s_O & 0 \\ d_Y & d_M & d_O & 1-r_D \end{pmatrix} \begin{pmatrix} a_Y \\ a_M \\ a_O \\ a_D \end{pmatrix}_t \tag{B8}$$

or

$$\mathbf{a}_{t+1} = \mathbf{M}\mathbf{a}_t. \tag{B9}$$

We based the natural mortality rates on published data from Lithuanian forests, setting $d_Y = 0.005$, $d_M = 0.007$ and $d_O = 0.009$ (after Ozolincius et al. 2005). By assuming an even-age structure within each age class we set the growth rates $g_Y = g_M = {}^1/_{35} = 0.0286$. We assume that deadwood persists for 10 years so that $r_D = 0.1$. Having set these parameters, we set the survival rates within each class such that the columns sum to 1, giving $s_Y = 0.9664$ ($= 1 - g_Y - d_Y$); $s_M = 0.9644$ ($= 1 - g_M - d_M$); and $s_O = 0.9910$ ($= 1 - d_O$).

Thus the matrix **M** projects changes in deciduous age-structure over time, for a fixed area of total forest, and its dominant eigenvalue has a value of 1.00. The associated left eigenvector has structure $\mathbf{a} = [0.213 \;\; 0.171 \;\; 0.544 \;\; 0.072]'$; therefore we re-assigned 11.7% [= 0.072/(0.072+0.544)] of our calculated initial values of A_O to the deadwood component A_D.

As the left eigenvector gives the long-term forest structure, we find that with weights of 0.25 for medium-age forest and 1.0 for old-age and deadwood, the forest structure approaches an overall habitat weight of 0.25(0.171) + 1.0(0.544+0.072) = 0.6582 (weighted hectares per hectare of deciduous forest). The exact trajectory that approaches this value depends on the initial structure of each county's deciduous forest. We therefore project the deciduous forest structure over time, finding the habitat value by applying equations B2 and B3 at each time-step. If all the productive forest area we consider here (Reese et al. 2003) were to be used for WBW habitat creation, the total number of WBW habitats in 200 years' time, assuming this simple model, would be just over 202.

The projection of forest structure under WBW conservation (Figure A1) indicates that the habitat score for each county i increases asymptotically from its initial value $h_{0,i}$ to some maximum, which we call $h_{i\,\max}$. We modelled the habitat score at time t therefore by

$$h_{t,i} = h_{i\max}\left(1 - \left(1 - \frac{h_{0,i}}{h_{i\max}}\right)e^{-\alpha_i t}\right) \tag{B10}$$

where α_i reflects how quickly the forest converges to $h_{i\,\max}$. We have estimated $h_{0,i}$ from data, and found $h_{i\,\max}$ by projecting the forest over 200 years (by which time the forest structure has become stationary). This curve is not an exact fit to our habitat projections, and so we estimated α_i (separately for each county i) by using the "LSQCURVEFIT" optimisation function in MATLAB (release 2009b, version 7.9.0; The Mathworks Inc.) to minimise the sum of squared errors in estimating α.

Our estimates of $h_{0,i}$ and $h_{i\max}$ reflect the total area of habitat productive forest in each county. For our economic analysis, however, it is more useful to consider the initial and maximum habitat in relation to forest area established as WBW conservation areas.

$q_{0,i} = h_{0,i} / \sum_x A_{x,i}$ (B11)

and

$q_{i\max} = h_{i\max} / \sum_x A_{x,i}$ (B12)

Using Equation B8 we can express the habitat improvement of a million-hectare conservation area established at time $t = 0$:

$$q_{t,i} = q_{i\max}\left(1-\left(1-\frac{q_{0,i}}{q_{i\max}}\right)e^{-\alpha_i t}\right) \quad \text{(B13)}$$

Because of the coarse nature of our data and the simplified forest projection model, it is necessary to include some degree of uncertainty in the model. We do this simply by allowing our estimates of $h_{0,i}$ and $h_{i\max}$ (and, hence, $q_{0,i}$ and $q_{i\max}$) to vary by ±50%.

Literature

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