



Ecological compensation of stochastic wetland biodiversity: National or regional policy schemes?

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ABSTRACT

The aim of this study is to compare policy schemes for ecological compensation applied at national and regional levels, using exploited inland wetlands as an example. We study whether uncertainty, due to natural variability and measurement difficulties, motivates compensation that is carried out in the same region as that of the exploited site, or whether it rather motivates nationwide compensation schemes. For this purpose, we develop an empirical, chance-constrained programming model of cost-effective wetland management. The model is spatially differentiated and accounts for heterogeneity in wetland quality across wetland types and regions. Wetland quality is defined by three alternative biodiversity indices: species richness, population-weighted species richness, and red-listed species richness, estimated from voluntarily reported data on breeding bird species observations. Results show that regional schemes are more expensive, in particular if the policy maker dislikes uncertainty and wants to prioritize uncommon species. Contrary to expectations from the theoretical analysis, regional schemes would lead to a higher risk-adjusted level of biodiversity at the national level. However, regionalization also implies that targets cannot be achieved if a high safety margin is imposed. Trading ratios are robust to the choice of wetland quality index.

1. Introduction

Human activities causing habitat destruction have led to a decline in natural terrestrial and aquatic ecosystems and pose a severe threat to many species worldwide (Millennium Ecosystem Assessment, 2005; Hansen et al., 2012; Seto et al., 2012). Wetlands are among the most sensitive on-land ecosystems. Over the last three centuries, the global wetland area has declined by 87%, and recent losses have been even more rapid (IPBES, 2019).

To counteract this process, policy makers in several countries have introduced requirements for compensation of the negative ecological impacts of economic development projects (Wissel and Wätzold, 2010; Briggs et al., 2009). In the United States, policy makers and private companies have developed policies that make use of market-like incentives for maintaining biodiversity and habitats, including wetland mitigation banks and conservation banks (Boisvert, 2015). These incentives can reduce costs for maintaining overall biodiversity (Wissel and Wätzold, 2010), and by enhancing the supply of conservation measures such banks could increase the likelihood that policy makers

require compensation measures from exploiters.

In the European context, compensation projects are usually designed and implemented specifically for each individual economic development project that is subject to such a requirement. This process is argued to result in high costs as well as uncertain and insufficient ecological outcomes (Briggs et al., 2009; Wätzold and Schwerdtner, 2005). In Sweden, where the present study is applied, compensation can be required under the framework of the Environmental Code when preventive damage mitigation measures are deemed insufficient. Compensation can then be relevant when areas with high natural values (e.g., in terms of biodiversity), green infrastructure, and important ecosystem services are impacted by exploitation. In addition, compensation can be required in the local context, then guided by the Planning and Building Act and typically focusing on the loss of natural environments close to urban areas, and areas of importance for local recreation. When compensation is required with support of the Environmental Code, the localization of the measures should focus on ecological functionality, but also consider costs as well as technical and practical conditions (SEPA, 2016). The Environmental Protection Agency's guidelines

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stipulate that it is often relevant to undertake compensation close to the place of the damage, e.g., in order to maintain local populations of endangered species and their dispersal opportunities (SEPA, 2016).¹ However, it is also acknowledged that the aims of compensation can sometimes be better achieved by implementation at a greater distance from the site of the exploitation, for the above-mentioned reasons (SEPA, 2016).

Several studies have analyzed the principles of ecological compensation and the role that economic considerations play for the optimal design of such policies. These studies show that there are economic and ecological efficiency gains from taking into account spatial connectivity, the timing of compensation measures in relation to that of exploitation, and uncertainty regarding the ecological effects of conservation and compensation measures (van Teeffelen et al., 2014; Wissel and Wätzold, 2010; Moilanen et al., 2009). It is shown that all these aspects can be dealt with through the use of trading ratios, implying that a greater compensation effort is required if the effect is uncertain, delayed, or implemented in a location with limited connection to similar habitats (Drechsler and Wätzold, 2009; Hartig and Drechsler, 2009; Wissel and Wätzold, 2010).

An issue of debate is whether compensation possibilities should be restricted with respect to the distance between the exploited site and the compensation site. On one hand, it is argued that such a restriction would reduce uncertainty about the ecological effect (van Teeffelen et al., 2014; Wissel and Wätzold, 2010), while simultaneously ensuring that local recreational values are maintained, given that the valuation of environmental services is typically declining in the distance to the site (Bateman et al., 2006). On the other hand, distance restrictions would reduce the cost-effectiveness of the scheme, as the possibility to make use of distantly located, low-cost measures with high biodiversity benefits is reduced (Wissel and Wätzold, 2010).

Few studies have carried out empirical economic analyses of ecological compensation. Using a landscape-level data set on 267 terrestrial vertebrate species, Polasky et al. (2005) developed a spatially explicit model for analysis of trade-offs between private returns to land and biodiversity outcomes, taking into account the role of habitat suitability and connectivity for the latter. Kangas and Ollikainen (2019) developed a dynamic model studying the ecological compensation of forest habitats. Uncertainty was addressed through Monte Carlo simulations that were used to identify the size of risk premiums on the market for ecological compensation. None of those has studied how the design of compensation should be adapted to the presence of uncertainty. In contrast, other studies have investigated the role of uncertainty for decisions on habitat conservation policies. Among those, Mallory and Ando (2014) applied Modern Portfolio Theory (MPT), originally developed by Markowitz (1952, 1959), to identify combinations of wetlands that should be chosen when the policy maker's objective is to minimize uncertainty about the benefits for a given level of expected benefits. In their application, uncertainty arises due to the limited knowledge about future impacts of climate change on the value of both private land and wetland biodiversity. Shah et al. (2017) developed this approach by using a two-stage procedure, which permits further spatial disaggregation of the results, but at the expense of possibly suboptimal outcomes. Also, drawing on chance-constrained programming (CCP) methods originally suggested by Charnes and Cooper (1959, 1963),

¹ The Environmental Code applies, e.g., for Natura 2000 sites. For those, guidelines stipulate that if possible, compensation measures should be carried out in the vicinity of the area where the damage occurs. However, and based on EU regulations, compensation measures could be implemented at larger distance, as long as they are carried out in the same biogeographical region in the same Member State (SEPA, 2016). Most of the area in Sweden belongs to the same biogeographical region, the boreal one. Only the southernmost part (Scania and part of Halland and Blekinge county) belongs to the continental region.

Gren et al. (2014) developed a dynamic model for cost-efficient forest habitat restoration under conditions of stochastic growth in habitat quality, and they derived trading ratios among regions and across time, but abstract from spatial covariation in habitat quality.

The aim of this paper is to model and compare national and regional schemes for ecological compensation when wetland quality is stochastic and correlated across space. We modelled wetland habitats in the agricultural landscape because wetlands are a common habitat in biodiversity compensation schemes both in Sweden and elsewhere (Blicharska et al., 2022; Josefsson et al., 2021). We collected data on wetland bird diversity in large agricultural wetlands (> 10 ha), using those to create measures of wetland quality. Wetland birds are the most common taxon considered in Swedish wetland compensation projects (Blicharska et al., 2022).² Additionally, wetland birds are commonly the focus of wetland conservation interventions such as the creation of new, and restoration of old, agricultural wetlands in Sweden (Aklilu and Elofsson, 2021; Kačergytė et al., 2021; Blicharska et al., 2022). Hence, the use of wetland bird diversities as a measure of wetland quality gives a high level of realism in our modeling of different biodiversity compensation scenarios.

Uncertainty about wetland quality is assumed to arise due to natural annual variability in the bird community as well as measurement difficulties. In particular, we study whether spatially correlated uncertainty is a motive for requiring that ecological compensation is only carried out in the neighborhood of exploited sites, or if the opposite applies. For this purpose, we develop an empirical chance constrained cost-effectiveness model of wetland management in Sweden. The model is spatially disaggregated and accounts for heterogeneity in wetland quality (i.e., wetland bird diversity) within and across regions, and for heterogeneity in opportunity costs of wetlands. To test the robustness of results, we compare outcomes for three alternative measures of stochastic wetland quality: wetland bird species richness, population-weighted bird species richness (giving rare species a higher weight), and red-listed bird species richness.

Results include a comparison of regional and national schemes with respect to costs, aggregate wetland quality, and trading ratios for exploited, restored, and constructed wetlands in different counties, and we examine the role of uncertainty for these outcomes. The paper contributes to the literature on ecological compensation through analysis of ecological compensation under spatially correlated uncertainty, and by empirical modeling of ecological compensation of wetland habitat.

The paper is organized as follows: in Section 2 we outline a stochastic model of ecological compensation, Section 3 describes the data, Section 4 reports the results, and Section 5 provides a discussion and conclusions.

2. A stochastic model of ecological compensation

Consider a country where wetland biodiversity is seen as valuable. However, demand for exploitation of land constitutes a challenge to the conservation of this biodiversity. Moreover, wetland biodiversity is subject to natural variability and can be difficult to measure. In order to understand the role of stochastic wetland biodiversity for policy, a simple model is developed in the following. For the purpose of this model, we assume that wetland quality is evaluated based on the biodiversity that it provides.

Assume that there are $i = 1, \dots, k$ wetlands in $j = 1, \dots, n$ regions contributing to aggregate wetland quality. The regions differ with respect to climate as well as land use. In each of the regions, existing wetlands can be restored, and new wetlands can be constructed to increase aggregate wetland quality.

More specifically, the aggregate wetland quality in a particular re-

² Losses of very small wetlands (<5 ha) sometimes also involve the compensation of amphibian species projects (Blicharska et al., 2022).

gion j depends on the supply of different wetland types, high-quality, medium-quality, and created wetlands, in the region. Let the variable q_{ijd} denote the quality of wetland i of type d in region j . The index for the type, with $d = HI, MED, CRE$, denotes whether the wetland is an existing one of high, HI , or medium, MED , quality or is newly created, CRE . The quality of a given wetland is assumed to depend on initial quality, q_{ijd}^0 , where we assume $q_{ij,HI} > q_{ij,MED} > q_{ij,CRE} = 0$ (i.e., the quality of created wetlands is assumed to be zero before the creation), and on measures undertaken to improve wetland quality, denoted m_{ijd} . The measure m_{ijd} can be thought of as restoration or creation of habitats, and it indicates the share of the wetland area subject to the measure in question, that is, we have $0 \leq m_{ijd} \leq 1$. The quality of a given wetland is then described by:

$$q_{ijd} = q_{ijd}(m_{ijd}; q_{ijd}^0). \tag{1}$$

We assume that aggregate quality of wetlands of type d in region j , b_{jd} , is found by summation of the quality across all wetlands of that type in the region. However, the aggregate wetland quality cannot be determined with certainty. Uncertainty arises because of natural variability and difficulties to accurately measure wetland quality. The stochastic processes are captured by an additive error term, ε_{jd} , and the stochastic wetland quality b_{jd} is thus defined by:

$$b_{jd} = \sum_i q_{ijd}(m_{ijd}; q_{ijd}^0) + \varepsilon_{jd}. \tag{2}$$

We assume that the function in Eq. (2) is continuous, differentiable, and concave. Aggregate wetland quality in the whole country, B , can then be expressed as the sum over wetland quality across regions and wetland types:

$$B = \sum_j \sum_d b_{jd}, \tag{3}$$

The aggregate national wetland quality is subject to uncertainty and the expected nationally aggregated wetland quality is then:

$$E(B) = E\left(\sum_j \sum_d b_{jd}\right) = E\left(\sum_i \sum_j \sum_d q_{ijd}\right), \tag{4}$$

while the variance of national aggregate wetland quality can be expressed as (see, e.g., Hogg and Craig, 1995):

$$Var(B) = Var\left(\sum_j \sum_d b_{jd}\right) = \sum_j \sum_d Var(b_{jd}) + \sum_j \sum_d \sum_{e \neq d} Cov(b_{jd}, b_{je}) + \sum_j \sum_{k \neq j} \sum_d Cov(b_{jd}, b_{kd}) + \sum_j \sum_{k \neq j} \sum_d \sum_{e \neq d} Cov(b_{jd}, b_{ke}). \tag{5}$$

From Eq. (5), we can see that the variance of national aggregate wetland quality consists of four different components: (i) the variance for each separate region and wetland type, reflected in the first term; (ii) the covariance between wetlands of different types in a given region due to, for example, similar climatic conditions and spatial connectivity, reflected in the second term; (iii) the covariance across regions for given wetland type, which could be due to habitat type factors, for example, valuable wetlands in different regions could share similar habitat characteristics that make them different from less valuable wetlands, which constitutes the third term; and (iv) the remaining covariance across wetland types and regions, which could be due to nationally common factors such as national variations in weather and data collection effort.

For each wetland category, there is a certain number of wetlands in a region, denoted n_{jd} , with $0 < n_{jd} \leq \bar{n}_{jd}$. Moreover, there is assumed to be

a characteristic, fixed size of wetlands of a given type in a given region, \bar{a}_{jd} . This is a simplification, given that wetland size could be related to the biodiversity provided, for example, to the number of bird species. The nature of the relationship between area and number of bird species is a contested issue, where conservation agencies tend to favor the protection of large, contiguous areas, whereas many meta-studies find that the number of species tends to be larger for multiple small wetlands, together covering the same total area (Fahrig, 2020). The above simplification is motivated by our focus on stochastic interdependences between different wetland types, and across space, rather than the best design of the individual wetland.

Wetland quality is assumed to be associated with a cost, $c_{jd} = c_{jd}(m_{jd}, n_{jd}; \bar{a}_{jd})$, reflecting the opportunity cost of abstaining from exploitation, and the cost for measures to improve wetland quality or create habitats on a given plot of land. The cost function is assumed to be convex, increasing, and differentiable.

The assumed decision problem of a national environmental agency is to determine a policy that minimizes the total cost, that is:

$$Min \sum_j \sum_d c_{jd}(m_{jd}, n_{jd}; \bar{a}_{jd}), \tag{6}$$

subject to a restriction $B \geq B^*$, which requires that aggregate national wetland quality exceeds a politically determined level, B^* . Several authors have proposed that it can be unsuitable to use the costs of expected wetland quality to evaluate cost-effectiveness, because ecological outcomes are frequently influenced by the variability and other aspects of the distribution (Mallory and Ando, 2014; Hoekstra, 2012; Shah and Ando, 2015; Shah et al., 2017). Policy makers could be concerned about variability if the benefits are nonlinear in wetland quality, or because they are risk averse. Therefore, the goal of the policy maker might better be expressed as the probability of achieving an aggregate wetland quality target. For example, if wetland quality should be larger than or equal to B^* with probability α , then this may be written as:

$$P\{B \geq B^*\} \geq \alpha,$$

$$\alpha \in (0, 1).$$

If $\alpha = 0.9$, this means that at least nine times out of ten, wetland quality must be greater than B^* . The deterministic equivalent of the above can be written as:

$$E(B) - K_\alpha \sqrt{Var(B)} \geq B^* \tag{7}$$

(see, e.g., Charnes and Cooper, 1959, 1963). The formulation in Eq. (7) implies that the stochastic aggregate wetland quality in the constraint is replaced by estimates of its values such as given by the expected value minus the quantity $K_\alpha \sqrt{Var(B)}$. The parameter K_α can be interpreted as the subjective weight that policy makers attach to standard deviation of wetland quality B . The higher is α , the larger is K_α , and the higher expected wetland quality is required to reach the same environmental target. Therefore, the larger is α , the larger is the cost for compliance with the wetland quality target. If $K_\alpha = 0$, policy makers do not attach any weight to variations in aggregate wetland quality, and the above can be interpreted as a deterministic constraint. The difference in minimum costs between the deterministic and chance-constrained outcomes depends on the chosen level of α , assumptions about the distribution of wetland quality, and the estimated $Var(B)$. Here, the total B is

assumed to be log-normally distributed, with:

$$B \sim \Lambda \left(e^{\mu_B + 0.5\sigma_B^2}, e^{2\mu_B + \sigma_B^2} (e^{\sigma_B^2} - 1) \right)$$

where Λ stands for the log-normal distribution (Greene, 1993). This implies that $\ln(B)$ is normally distributed with $\ln(B) \sim N(\mu_B, \sigma_B^2)$. An advantage of using the log-normal distribution compared to a normal distribution, such as used in, for example, Gren et al. (2014), is that it excludes the possibility of a negative total wetland quality. Given the probability requirement α and mean and variance of total loads, the critical values for the log-normal distribution, $K_{ln\alpha}$, can be written in terms of the critical values of the normal distribution (Gren et al., 2002).

2.1. Cost-effective allocation of restoration and construction measures

The optimal allocation of restoration and construction measures is affected by the choice of probabilistic load targets such as in Eq. (7). This can be seen when solving the cost minimization problem for the policy maker with respect to the different measures. The cost minimization problem can be written as:

$$\text{Min}_{n_{jd}, m_{jd}} \sum_j \sum_d c_{jd} (m_{jd}, n_{jd}; \bar{a}_{jd}) \tag{8}$$

s.t. (1)–(5), (7) and

$$0 \leq n_{jd} \leq \bar{n}_{jd}.$$

$$0 \leq m_{jd} \leq 1.$$

The objective function is convex according to assumptions made about cost functions. It is assumed that expected wetland quality is increasing and quasi-concave in measures, and the variance-covariance matrix for wetland quality is positive semi-definite by definition. Together with the assumption that $K_{ln\alpha} > 0$, this ensures that the probabilistic constraint in Eq. (5) for the cost minimization problem is quasi-concave (Paris and Easter, 1985). Thereby, the Kuhn-Tucker conditions are sufficient for defining a unique solution to the cost minimization problem (Takayama, 1993). The cost minimizing level of the different

$$\frac{1}{2} K_{ln\alpha} [Var(B)]^{\frac{1}{2}} \frac{\partial Var(B)}{\partial n_{jd}} = \frac{1}{2} K_{ln\alpha} [Var(B)]^{\frac{1}{2}} \left[\sum_j \sum_d \frac{\partial Var(b_{jd})}{\partial n_{jd}} + \sum_j \sum_d \sum_{e \neq d} \frac{\partial Cov(b_{jd}, b_{je})}{\partial n_{jd}} + \sum_j \sum_{k \neq j} \sum_d \frac{\partial Cov(b_{jd}, k)}{\partial n_{jd}} + \sum_j \sum_{k \neq j} \sum_d \sum_{e \neq d} \frac{\partial Cov(b_{jd}, b_{ke})}{\partial n_{jd}} \right]. \tag{14}$$

abatement measures are given by the Kuhn-Tucker conditions (10–11):

$$L = \sum_j \sum_d c_{jd} (a_{jd}, m_{jd}, n_{jd}) - \lambda [E(B) - K_{ln\alpha} \sqrt{Var(B)} - B^*] + \mu_{jd} (n_{jd} - \bar{n}_{jd}) \tag{9}$$

$$\frac{\partial c}{\partial n_{jd}} - \lambda \left\{ \frac{\partial E(B)}{\partial n_{jd}} - \frac{1}{2} K_{ln\alpha} [Var(B)]^{\frac{1}{2}} \frac{\partial Var(B)}{\partial n_{jd}} \right\} + \mu_{jd} \geq 0, n_{jd} \geq 0 \tag{10}$$

$$\frac{\partial c}{\partial m_{jd}} - \lambda \left\{ \frac{\partial E(B)}{\partial m_{jd}} + \frac{1}{2} K_{ln\alpha} [Var(B)]^{\frac{1}{2}} \frac{\partial Var(B)}{\partial m_{jd}} \right\} + \omega_{jd} \geq 0, m_{jd} \geq 0 \tag{11}$$

In the above equations, λ denotes the Lagrange multipliers of the chance constraint on aggregate wetland quality. The terms μ_{jd} and ω_{jd} are the shadow values of the capacity constraints on wetland number and measures to improve wetland quality, respectively. The conditions in Eqs. (10) and (11) state that the marginal cost in optimum of each measure is equal to or larger than the measure's marginal impact on the environmental target. The complementary slackness conditions reveal that if the marginal cost exceeds the impacts on the target, then the use of that measure must be zero in the optimum. Conversely, for a measure included in the optimal solution, the marginal cost equals the value of

the impact on the targets.

Assuming an interior solution, the trading ratio between wetlands of type d in region j , and type $e \neq d$, in region $k \neq j$, is defined by:

$$\frac{\frac{\partial c}{\partial n_{jd}} + \mu_{jd}}{\frac{\partial c}{\partial m_{ke}} + \omega_{ke}} = \frac{\frac{\partial E(B)}{\partial n_{jd}} - \frac{1}{2} K_{ln\alpha} [Var(B)]^{\frac{1}{2}} \frac{\partial Var(B)}{\partial n_{jd}}}{\frac{\partial E(B)}{\partial m_{ke}} - \frac{1}{2} K_{ln\alpha} [Var(B)]^{\frac{1}{2}} \frac{\partial Var(B)}{\partial m_{ke}}} \tag{12}$$

Hence, in optimum, if the trading ratio in (12) exceeds one, this means that more than one unit of wetland in region k of type e (i.e., n_{ke}) should be traded against one unit of wetland in region j of type d (i.e., n_{jd}) in order to maintain a given level of B^* , as defined by Eq. (7). Further, the trading ratio between wetland area and measures to improve wetland quality is defined by:

$$\frac{\frac{\partial c}{\partial n_{jd}} + \mu_{jd}}{\frac{\partial c}{\partial m_{ke}} + \omega_{ke}} = \frac{\frac{\partial E(B)}{\partial n_{jd}} - \frac{1}{2} K_{ln\alpha} [Var(B)]^{\frac{1}{2}} \frac{\partial Var(B)}{\partial n_{jd}}}{\frac{\partial E(B)}{\partial m_{ke}} - \frac{1}{2} K_{ln\alpha} [Var(B)]^{\frac{1}{2}} \frac{\partial Var(B)}{\partial m_{ke}}} \tag{13}$$

Thus, Eq. (13) shows the trading ratio between applying the measure m in region k on wetlands of type e , against one unit of wetland in region j of type d .

2.2. Marginal impact of measures on expected wetland quality and variance thereof

Using Eq. (12), the impact of a measure on the target can be divided into two components: the marginal change in expected aggregate wetland quality, $\frac{\partial E(B)}{\partial n_{jd}} \geq 0$, and the marginal change in the variance of wetland quality, $\frac{1}{2} K_{ln\alpha} [Var(B)]^{\frac{1}{2}} \frac{\partial Var(B)}{\partial n_{jd}}$. If the latter term is negative for a particular type of wetland, the trading ratio in optimum is higher than if it is positive. Thus, exploitation of a wetland that reduces the variability of overall wetland quality requires a larger area of other wetlands in compensation when variability is considered by the decision-maker.

The expression for the marginal impact on variance can also be expanded to illustrate the role of covariance between wetlands. When expanding, one has that:

The first term in Eq. (14) is positive, whereas the three following terms are positive if covariance is positive and negative if covariance is negative. One can think of different possible situations: a wetland may have a small positive impact on aggregate regional variance due to small variability of wetland quality on that particular wetland type, but it may have a large positive impact on the sum of the covariance terms if wetland quality for this type of wetland is positively correlated with wetland quality of other wetland types in the same region and with the quality of wetlands in other regions. Alternatively, a wetland could have a large positive impact on variance of regional wetland quality, because the quality of that particular type of wetland is highly variable, but it could have a small positive impact on total variance if the quality of this particular wetland type is negatively correlated with much of the quality of other wetland types and regions. Thus, for example, if wetlands are created for which the wetland quality is negatively correlated with that of other wetlands, fewer such wetlands are necessary to compensate a loss of other wetlands, ceteris paribus.

As mentioned in the introduction, targets for conservation might be set at the national or regional level. Eq. (7) expresses a target for the national level, summarizing outcomes across all regions j . If targets for

wetland quality conservation are instead set separately for each region j , trading ratios can be identified for wetlands of different type in the same region, and for measures, m_{jd} , versus wetland number. However, if targets are set for the regions j , the covariance in wetland quality across regions will be ignored, that is, the two last terms in Eq. (14) will not appear in the constraints for any of the regions. Then, if covariance across regions is positive (negative), this implies that less (more) effort is necessary to meet the regional probabilistic constraints. Hence, with regional probabilistic constraints, and positive correlation in wetland quality, the total certainty equivalent wetland quality at the national level could be lower compared to the case with a national target, because some of the stochasticity is ignored.

3. Data

3.1. Ecological data

We use spatially differentiated data for our analysis. In our model, the smallest spatial unit is the county level, and there are 21 counties in Sweden. In addition, Sweden is divided into three larger regions, Götaland, Svealand, and Norrland, which differ with respect to climatic conditions.

Wetlands are important habitats for birds in all parts of Sweden, and birds are one of the most commonly monitored species groups, which allows for comparisons across wetlands. In general, biodiversity values (i.e., wetland quality) can be measured in many different ways. The two most common biodiversity estimates used in ecological compensation studies are species richness and abundance of species (i.e., number of individuals of species; Josefsson et al., 2021). Although a full compensation of wetland loss should admittedly include the loss of species' local population sizes, abundance is usually only considered when the goal is to compensate the loss of one or a few species. When the goal is to compensate for the loss of a whole community of species, species richness is typically the measure used (Josefsson et al., 2021), which motivates our choice to use wetland bird species richness in order to measure wetland quality.

The species richness estimates are based on voluntarily collected data on observations on breeding bird species from 60 wetlands in Sweden over the years 2005 to 2014, obtained from the Swedish Species Observation System (Artportalen, www.artportalen.se).³ The use of this dataset is motivated by the high data coverage of different locations throughout the country, which is useful when the purpose is to analyze ecological compensation across spatial scales. These wetlands are spread throughout southern Sweden and along the northern coast.⁴ The wetland quality indices are estimated using an occupancy modeling framework that includes different detection probabilities of species, following the approach in Ruete et al. (2017). We impose the restriction that a bird species must be present at least 20 days during April–June at a given wetland to be included in the data as a breeding species. For the dataset, we distinguish between high-quality and medium-quality wetlands by assuming that 25% of the wetlands in the dataset, those with the highest level of the wetland quality indices, are of high quality, while the remaining ones are assumed to represent medium quality.

We used three richness indices to measure wetland quality: species

³ A criterion for the selection of wetlands to include was that bird observations could be linked to a specific, established wetland area for which the relevant given size of the wetland area could be determined.

⁴ 40 of these wetlands have at least some area that belongs to a Natura 2000 site. Five of those are located in the south, belonging to the continental biogeographical region, while the other 35 are located in the boreal region. In our calculations we do not account for Natura 2000 status, as our model mainly strives to illustrate the implications of stochasticity of biodiversity for biodiversity offsetting, and the purpose of the empirical analysis is to illustrate these mechanisms with relevant data.

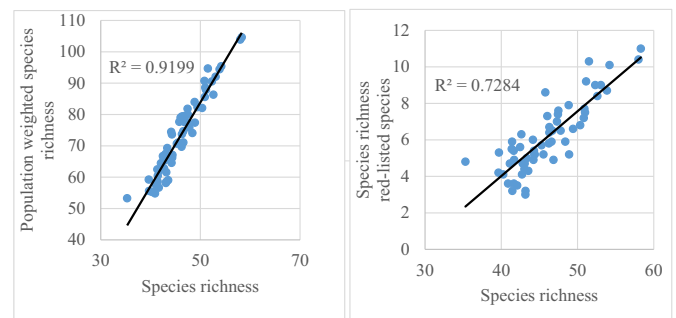


Fig. 1. Relationship between species richness and population-weighted species richness (left) and red-listed species richness (right) for data 2005–2014 across 60 wetlands. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

richness, population-weighted species richness, and red-listed species richness. This choice of richness measures can help to understand how compensation schemes are affected by applying biodiversity measures with different focus on common versus rare species. The bird data is used to calculate expected wetland quality, and variability thereof, for different wetlands categories in the three large regions for the three measures of wetland quality. The measure of species richness estimates the total number of breeding wetland bird species in each wetland⁵ (see species list of included species in the Supplementary Material, Table S1). Species richness is often used as a simple and straightforward measure of biodiversity values of wetlands, and it is a common measure used in conservation evaluations. However, simply calculating species richness risks presenting problems because all species are assumed equally valuable. In practice, species are valued differently by researchers, policy makers, and the general public (Hiron et al., 2018), exemplified by the globally widespread use of red lists, which ranks species on a scale from being without threats to being endangered (Rodrigues et al., 2006). Therefore, we also investigated two indices of richness that take into account that species differ in their conservation values. To address this issue, we calculated population-weighted species richness, which is a measure of wetland quality giving more weight to nationally rare bird species. This measure is based on a weighting against mallard (*Anas platyrhynchos*), which is the most common wetland species with a total of about 200,000 pairs in Sweden. The weighting was calculated as the logarithm of the ratio between the mallard population and the population of the species in question according to population estimates in 2012 (Ottosson et al., 2012). For example, the weighted value of the Black tailed godwit (*Limosa limosa*), which had an estimated population of 75 pairs, was $\log(200,000/75 + 1) = 4.43$. That means that the presence of a black tailed godwit corresponds to the presence of about four common species. These species-specific, weighted values were then summed for each wetland and year. In addition, we calculated the richness of red-listed species to give full weight to species of conservation concern as ecological compensation need to especially account for the loss of such species. For this measure, the number of species in the three threat categories “acutely endangered, highly endangered and vulnerable” (Swedish Species Observation System, 2015) were summed up. Overall, the species richness, population-weighted species richness and red-listed species richness indices are highly correlated, although the relationship is weaker between the species richness and red-listed species richness indices (Fig. 1).

Using the three different richness indices as measures of wetland

⁵ Passerines were excluded because passerines are (i) not a target taxon in Swedish wetland conservation projects, including restoration and creation of wetlands, (ii) not declining in numbers, probably because many species benefit from agricultural wetlands getting overgrown by reeds and shrubs (Kačergytė, 2021).

Table 1
Expected wetland quality per wetland, coefficient of variation of regional wetland quality, and number and area of wetlands for three wetland quality indices.

Region	Type	# of wetl. Type	Area per wetl. (ha)	Bird species richness		Population-weighted bird species richness		Red-listed species richness	
				Exp. wetland qual. Per wetl.	Coeff of var. regional wetland qual.	Expected wetland qual. Per wetl.	Coeff of var. regional wetland qual.	Expected wetland qual. Per wetl.	Coeff of var. regional wetland qual.
Götaland	High qual.	11	817	52.21	0.022	90.71	0.035	8.73	0.056
	Med. qual.	11		45.52	0.027	69.78	0.038	5.30	0.071
Svealand	High qual.	4	476	53.03	0.034	91.26	0.047	9.20	0.095
	Med. qual.	27		43.25	0.021	66.16	0.028	5.15	0.091
Norrland	High qual.	0	502	NA	NA	NA	NA	NA	NA
	Med. qual.	7		43.83	0.029	68.48	0.039	5.61	0.098

quality, we can calculate expected (mean) wetland quality, and its coefficient of variation,⁶ for high-quality and medium-quality wetlands in different regions (see Table 1). Data on expected wetland quality for a given region is assumed to apply for all counties within that region. The coefficient of variation of wetland quality expresses the variability of aggregate wetland quality for wetlands of a given type in a given region, i.e., expresses the variability of b_{jd} . Moreover, we use the same dataset to calculate representative wetland size, \bar{a}_{jd} , for wetlands in different regions. For this calculation, we do not distinguish between wetland types because of the limited number of wetlands in some cases (see Table 1). Comparing the different wetland quality indices so obtained, one can note that both the ratio $E(b_{IH})/E(b_{IMED})$ and the coefficients of variation are the lowest for species richness, and the highest for red-listed species richness. One can note that we have not attempted to estimate the “true” probability distribution.

For created wetlands, we use data from bird surveys carried out during the breeding season in the year 2018 in such wetlands in Uppsala county in mid-east Sweden for the calculation of expected wetland quality and variability thereof. The same set of bird species as for the wetlands in Table 1 was inventoried twice from mid-May to early June. The criterion for a species to be noted as present was that it was observed in at least one of the two visits, in combination with breeding signs in the field. Hence, migratory species and molting individuals were excluded. In total, 89 created wetlands were surveyed, walking at a steady pace around the wetland (10 min/ha, line transects) and counting all observed birds and their breeding behavior within the wetland and 50 m around the shore. For a more detailed description of the field methods, see Kačergytė et al. (2021).

Based on this dataset, the expected species richness, population-weighted species richness, and red-listed bird species richness on created wetlands was estimated to 6.92, 13.02, and 0.12, respectively, per created wetland. Comparing to Table 1, this suggests that created wetlands would be relatively efficient in increasing population-weighted species richness, and relatively inefficient in increasing red-listed species richness. We assume that the mentioned data apply to created wetlands in Svealand region, where Uppsala county is located. For Götaland and Norrland, we obtain the corresponding figures for created wetlands by weighing the numbers for Svealand by the ratio of the expected quality of medium-quality wetlands in the regions in question. Based on the same data, we calculate the coefficient of variation for created wetlands, which then ranged between 0.7 and 0.8 for the three indexes. This suggests that the *ex ante* uncertainty regarding the biodiversity that can be achieved at created wetlands is about an

order of magnitude larger than the uncertainty that relates to the biodiversity on existing wetlands, where the latter is reflected in the coefficients of variation reported in Table 1. For our analysis, we assume that the coefficients of variation for created wetlands are ten times the highest coefficient of variation for the relevant wetland quality index as reported in Table 1.⁷

We use data from the Board of Agriculture for the years 2015 to 2017 on wetlands created with financial support through the Rural Development Program in order to obtain data on the representative area for created wetlands in different counties. These areas are calculated as the average area per county over the three years. The representative area then ranges from 2.4 to 21.6 ha for the different counties (see Table A1 in the Appendix). Wetlands were created in all counties except in the counties in Norrland. We assume that if wetlands are created in Norrland, they have a size of 5 ha.

Furthermore, we calculated correlation coefficients⁸ for aggregate quality for different wetland types and geographical regions, using the same wetland dataset as for Table 1. Thus, we calculate the correlation coefficients for the variable b_{jd} , using data for the years 2005 to 2014 (see Tables A2a–A2c in the Appendix). For created wetlands, we somewhat arbitrarily assume that the correlation with other wetland types equals 0.95 for all types and regions, which is similar to the highest correlation found among existing wetlands. The assumption is motivated by the aim to take a conservative approach toward the introduction of constructed wetlands, due to the potentially higher uncertainty about their prospects as a suitable habitat for bird species.

Restoration is assumed to only be undertaken on wetlands with medium quality, and the effect is assumed to be an increase of wetland quality to the level found for wetlands with high quality in the same region. For regions with initially zero wetlands in the high-quality category, it is assumed that restoration is not feasible.

3.2. Economic data

Conservation of existing wetlands and creation of new wetlands is assumed to be associated with an opportunity cost, considering that the land cannot be used for agricultural purposes. The opportunity cost of agricultural land is taken to be equal to the lease price for agricultural land, obtained from SCB (Statistics Sweden) (2019a); see Table A1 in the

⁶ The coefficient of variation is calculated as $\sqrt{\text{Var}(b_{jd})}/E(b_{jd})$, using data from 2005 to 2014.

⁷ I.e., for the species richness index, the coefficient of variation for created wetlands is assumed to be 0.340, while for the population-weighted species richness index and the red-listed species richness index it is assumed to be 0.470 and 0.950, respectively.

⁸ The correlation coefficient is calculated as $\text{Cov}(b_{jd}, b_{ke}) / (\sqrt{\text{Var}(b_{jd})} \sqrt{\text{Var}(b_{ke})})$, using data from 2005 to 2014.

Appendix. In addition, wetlands require maintenance to sustain their quality over time. We assume that the maintenance cost is equal for all wetlands and equals the agri-environmental support to wetland maintenance, 4000 SEK/ha. Creation and restoration of wetlands is also associated with a cost. We calculate this cost based on data in Flyckt (2010), reporting costs for six restoration projects across different locations in Sweden and implemented between 1999 and 2009, and in Österling and Kindt (2007), reporting costs for 85 projects implemented in Laholm municipality in south Sweden between the years 2001 and 2005 (see Table S2 and S3 in the Supplementary Material, respectively). We calculate the annuity of the reported costs using a 3% discount rate and assuming a project lifetime of 30 years. Costs differ considerably across reported projects and between studies. Average per hectare costs in Flyckt (2010) are >6 times larger than in Österling and Kindt (2007). Given that there are two outliers in Flyckt (2010), with much higher annuity costs per hectare than other projects, we chose to use the median per hectare costs across all projects in the two studies as a representative measure of the costs. The resulting annual cost is then 7206 SEK per hectare and year, which is assumed to apply for both restoration and creation.

For the analysis, we also need information on initial and potential wetland areas. The total initial area of highly and medium valuable wetlands is assumed to equal the total area of wetlands in our dataset for Table 1, which is 36,246 ha. We assume that the wetland area of a given type (high or medium) in a given county is proportional to the wetland area of that type in the region (see Table 1), adjusted for the county's area of agricultural land compared to the area of agricultural land in the region. The area of agricultural land was calculated as the area of arable and grazing land in 2015, obtained from Statistics Sweden; see Table A1. The number of high and medium valuable wetlands is treated as exogenous in the calculations: the number of medium valuable wetlands is assumed equal to the initial number, and the number of high valuable wetlands is assumed to be reduced by a given percentage, due to development projects.

We base our subjective estimate of the area where wetlands could be created on data on the agri-environmental support to wetland maintenance, where support was provided for approximately 9500 ha in the whole country in 2015 (BOA, 2018). About 80% of the applications for support to wetland maintenance were linked to support for wetland restoration and creation, confirming close links between restoration and creation on one hand, and maintenance support on the other. This area was distributed across counties, assuming that an equal number, 76 wetlands, can be created in each county.

4. Results

The results are calculated using the optimization software GAMS, version 23.9.2 (GAMS Development Corporation, 2019), and a CONOPT3 solver. We first estimate the total cost of maintaining aggregate national wetland quality for the different wetland quality indices, under conditions of uncertainty. Second, we examine differences between nationally and regionally implemented schemes for ecological compensation. This is done by comparing costs for meeting conservation targets at national and regional levels, respectively, calculating the resulting certainty equivalent wetland quality at the national level, and comparing the implications for the allocation of exploited, restored, and created area across counties. Finally, we identify trading ratios between exploited, restored, and created wetland area within and across different regions for national and regional compensation schemes, respectively.

4.1. Total costs as a function of required reliability of target achievement

We examine how the total cost changes with the required reliability of target achievement, when the target is set at the national level. For all wetland quality indices, the coefficients of variation in regional wetland quality for a given type of wetland are small, while correlation in quality

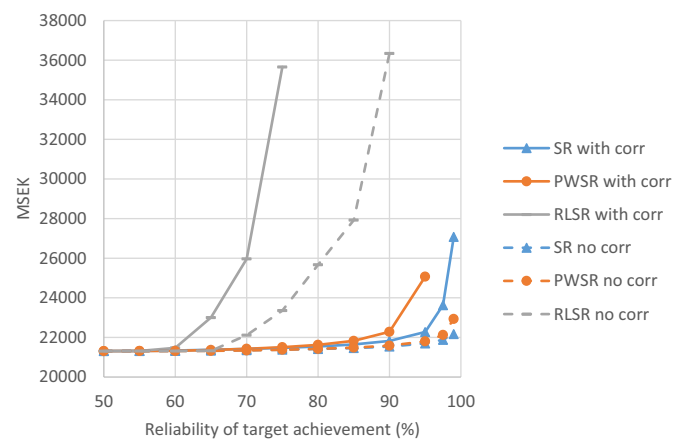


Fig. 2. Cost in Million SEK per year of maintaining national wetland quality when 15% of the high-quality wetlands are exploited, with and without consideration of correlation of quality across wetland types and regions. Costs include opportunity costs of agricultural land, and costs for wetland maintenance, restoration, and creation. SR = species richness, PWSR = population-weighted species richness, RLSR = red listed species richness. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

between different wetland types and regions is large and positive. To investigate the implications of these two different dimensions of uncertainty, we first include correlation with data as described above, then repeat the estimations with all correlations assumed equal to zero. In both cases, the cost of maintaining aggregate wetland quality is calculated for a 15% reduction in the area of high-quality wetlands in all counties (see Fig. 2). This implies a large absolute reduction in valuable wetlands in Götaland and no reduction in Norrland, which is similar to actual development in Sweden, where there is a higher degree of exploitation further south in the country.

Results show that costs are increasing in the required reliability level of target achievement. The costs increase more rapidly with the reliability level for the red-listed species index, while the slowest rate of increase is found for the species richness index. The main explanation is the relatively low provision of expected red-listed species richness on created wetlands, that are used to comply with the successively more stringent target, in combination with the comparatively high coefficients of variation for the red-listed species richness index. Correlation in quality across wetland types and regions accounts for a major share of total variability for the red-listed species already at low levels of reliability, and therefore has a considerable impact on the cost level. When correlation is included in the calculations, the cost for meeting the national target with 70% certainty is 52% higher for the red-listed species richness index, compared to the case when correlation is not included. For the other two indices, correlation only has a significant impact on policy costs when the reliability level is 90% or higher.

4.2. Regionally separate or national targets for maintaining aggregate wetland quality

Given the debate about whether ecological compensation should be permitted when the distance between the exploited location and the compensation site is large, we investigate the implications of spatially restricting compensation. We first compare costs for maintaining aggregate wetland quality using national and regional schemes for compensation. Similarly as above, we assume that highly valuable wetlands are reduced by 15% in all counties. The outcome is compared for 50%, 55%, and 60% reliability of target achievement (see Fig. 3). The

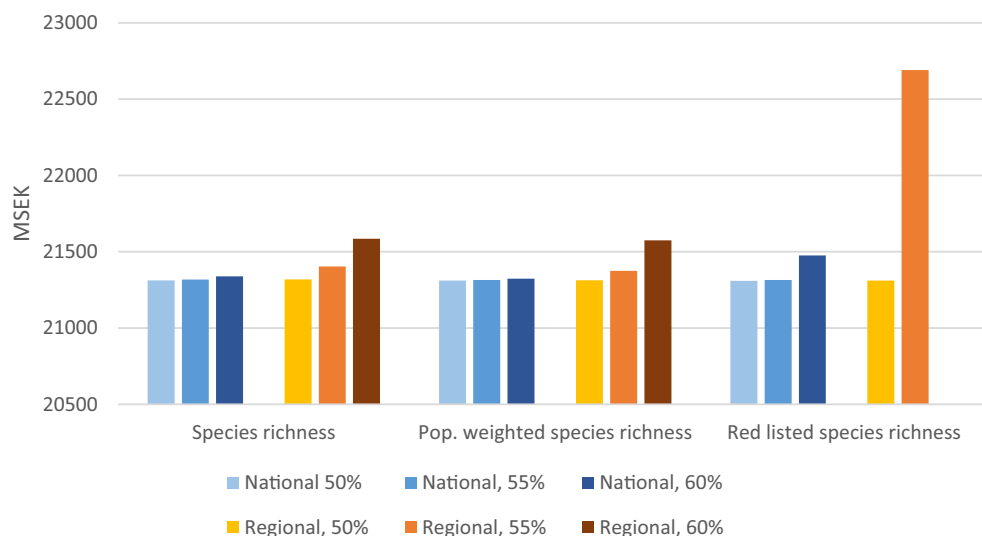


Fig. 3. Total costs in Million SEK per year of regionally separate and national targets, respectively, using different wetland quality indices, and with 50%, 55% and 60% reliability of meeting the no-net-loss target when 15% of the high value wetlands are exploited. Correlations are included. The regionally separate targets cannot be achieved for the red-listed species richness index under 60% reliability. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

reliability levels are chosen such that targets can be met in all cases for the two lower levels, and the national target can be met when using the species richness and population-weighted species richness indices.⁹

The results show that meeting regionally separate targets is more expensive than meeting national ones, as could be expected. The effect is small under risk neutrality (i.e., 50% reliability) but larger when uncertainty is considered at the 55% and 60% reliability levels. The additional cost under regionally separate targets is due to the need to restore and create wetlands in Götaland, where the loss of valuable wetland area is large in absolute terms and the opportunity cost of land is high. Regionally set targets are particularly expensive when the red-listed species index is applied and the target should be met with a higher reliability, which is explained by the larger coefficients of variation for this index, in combination with the low expected level of the red-listed species index for created wetlands. The former implies that the required safety margin is higher than for the two other indices, while the latter implies that the potential for increasing expected wetland quality through wetland creation is relatively smaller.

The use of regional targets has additional implications: it implies that correlation in wetland quality across regions is ignored. Given the positive correlations found across regions, this implies that there is a risk that regional targets lead to a lower certainty equivalent of the national aggregate wetland quality (i.e., a lower $E(B) - K_{\alpha}\sqrt{Var(B)}$; see Eq. (7)) than if the same reliability level was applied at the national level.

We therefore calculated the certainty equivalent of the aggregate wetland quality under regionally separate and national compensation schemes. Fig. 4 shows that regionally separate targets lead to an increase in the national certainty equivalent by 10.5%, 15.4%, and 14.9% for the three wetland quality indices, respectively, at 55% reliability of meeting targets. The smallest effect is found for the species richness index, and the population-weighted species richness index yields a slightly larger effect than the red-listed species richness index. The increase in the certainty equivalent under regionally separate targets is mainly due to an increase in expected wetland quality, caused by increased restoration of medium-quality wetlands in Götaland. This measure was not cost-effective under a common target. These wetlands have a quality that is less uncertain than the medium-quality wetlands in Svealand where much of the restoration is done under a common target. This illustrates the reduced possibilities to find low-cost measures, and effectively manage risk, under regionally separate targets, because the within-

region variation in costs and uncertainty among the available conservation options is small compared to the variation found at the larger spatial scale.

It could be argued that there is a risk that a national compensation scheme would imply that valuable wetland habitats could be lost in south Sweden, where the exploitation of land is high, and replaced by wetlands created in North Sweden where the opportunity cost of land is much lower. We investigate this issue considering a scheme applied at either national or regional level, using the population-weighted species richness index. In Fig. 5, we show the resulting distribution of exploited valuable wetlands, and restored and created wetlands, across counties and regions for 55% reliability of meeting national and regional targets, respectively. The figure shows that the total number of hectares, where restoration and creation is undertaken, is larger under a regional compensation scheme than under a national one. Under a national scheme, compensation for exploited wetlands in Götaland and Svealand is carried out in Svealand, where the opportunity cost of land is lower and the effect of restoration is higher. Under a regional scheme, wetlands are restored and created also in Götaland to compensate for the habitat loss in the same region. In Norrland, there is, by assumption, neither losses of valuable habitat nor restoration carried out, independently of the target. Creation of wetlands in Norrland is small, and amounts to <4 ha in the regional solution.

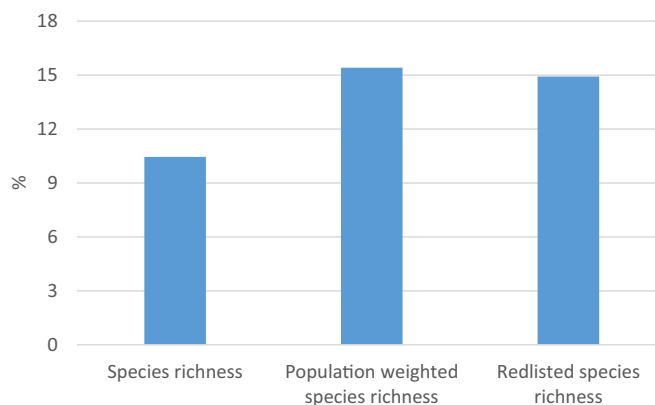


Fig. 4. Percentage increase in the certainty equivalent of national aggregate wetland quality under regionally separate targets compared to national targets, given 55% reliability of meeting targets.

⁹ The regional target for Norrland cannot be met under the red-listed species index and 60% reliability.

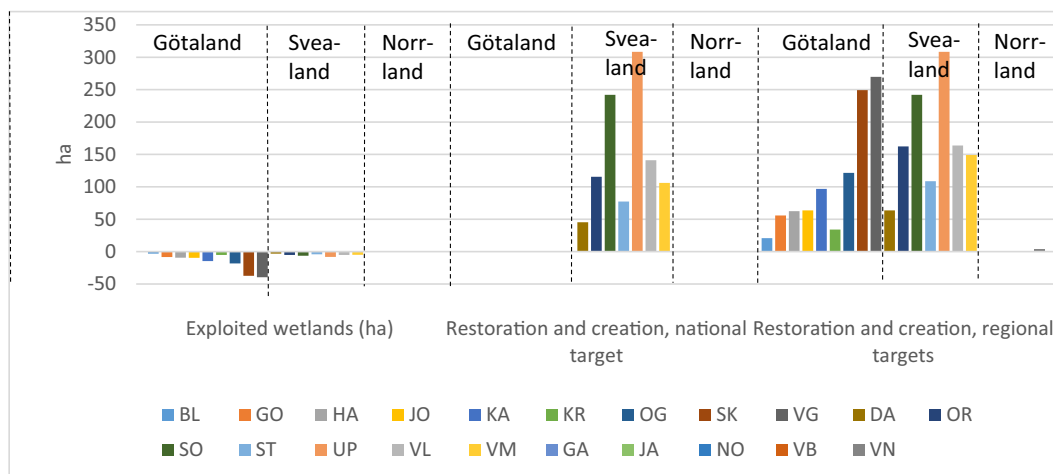


Fig. 5. Spatial distribution of exploited area (hectares) of valuable wetlands, restored wetlands under a national target, and restored and created wetlands under regional targets. The exploited area is 15% of the original area of highly valuable wetlands. Calculations are made for 55% reliability using the population-weighted species richness index. Creation of wetlands is small in magnitude and only occurs under regional targets, it is therefore aggregated together with restoration. The columns indicate counties: In Götaland: BL = Blekinge, GO = Gotland, HA = Halland, JO = Jönköping, KA = Kalmar, KR = Kronoberg, SK=Skåne, OG = Östergötland. In Svealand: DA = Dalarna, ST = Stockholm, SO=Södermanland, UP=Uppsala, VL = Värmland, VM = Västmanland, VG = Västra Götaland, OR = Örebro. In Norrland: GA = Gävleborg, JA = Jämtland, NO=Norrbottnen, VB=Västerbotten, VN=Västernorrland.

Table 2

Trading ratios, measured as the number of hectares restored, R, or created, C, to compensate for a loss of one hectare of high-quality or medium-quality wetlands. Trading ratios are calculated for a 15% reduction in the area of valuable wetlands and 55% certainty of meeting national and regional targets of preserving aggregate wetland quality.

			National target		Regional target							
			Svealand		Götaland				Svealand		Norrland	
Red. by 1 ha			R	R	R	R	C	C	R	R	C	C
			Min	Max	Min	Max	Min	Max	Min	Max	Min	Max
Species richness index	High	Götal.	1.49	2.11	1.48	2.10	7.51	10.66				
	Med.	Götal.	1.49	2.11	1.48	3.10	12.60	15.75				
	High	Sveal.	1.26	1.58					1.26	1.58		
	Med.	Sveal.	1.26	2.48					1.87	2.08		
	Med.	Norrl.	1.20	1.26							2.92	3.07
Pop. weighted species richness	High	Götal.	1.49	2.11	1.48	2.10	7.52	10.66				
	Med.	Götal.	1.49	2.11	2.48	3.10	12.61	15.76				
	High	Sveal.	1.49	1.59					1.58	1.60		
	Med.	Sveal.	1.49	2.07					1.80	2.60		
	Med.	Norrl.	1.26	2.00							2.93	2.93
Red-listed species richness	High	Götal.	1.49	2.11	1.42	2.08	6.76	9.72				
	Med.	Götal.	1.49	2.11	2.49	3.15	11.86	15.01				
	High	Sveal.	1.26	1.58					1.26	1.58		
	Med.	Sveal.	1.26	2.58					1.63	2.06		
	Med.	Norrl.	1.20	1.26							2.32	2.32

4.3. Trading ratios between measures

Finally, we calculate the trading ratio between compensation measures (i.e., wetland restoration and creation) and wetland area loss according to the lhs. of Eq. (13), including shadow values of the capacity constraints. For ease of interpretation, we recalculate the trading ratios obtained from Eq. (13) as the number of hectares of wetland restoration and creation necessary to compensate for a loss of one hectare of high-quality or medium-quality wetlands (see Table 2).¹⁰ Calculations are made for both national and regional targets for conservation of aggregate wetland quality, given a 55% reliability requirement, and using the

¹⁰ More specifically, the trading ratios presented in Table 2 are calculated as $(\frac{\partial c}{\partial \mu_{jd}} + \mu_{jd}) / (\frac{\partial c}{\partial m_{ke}} + \omega_{ke}) \bar{a}_{ke}^* / \bar{a}_{jd}$.

three different wetland quality indices.¹¹

A trading ratio expresses the rate of compensation in the cost-efficient solution, taking into account both the environmental impact and the costs of the measures. Therefore, the trading ratios will vary with both the impact of conservation, restoration, and creation on the aggregate wetland quality, and with the costs of these measures. In our model, wetland quality is differentiated at the regional and wetland type level, while the opportunity cost of land is differentiated at the county level, implying that trading ratios vary at the county level because this is the lowest level of spatial disaggregation in the data.

Table 2 shows the trading ratios for a loss of one hectare of high-quality or medium-quality wetland in a given region against restored and created hectares of wetland in the same and other regions. We show

¹¹ A regional scheme cannot be implemented in Norrland with >55% reliability.

the minimum and maximum values of the trading ratios across regions. Trading ratios for restoration and creation measures are only included in the table when these measures are included in the cost-effective solution. One can note that under a national target, more restoration is required in Svealand to compensate for high-quality wetland loss in Götaland, than for high-quality wetland loss in Svealand, even though the expected wetland quality loss for high value wetlands is higher in Svealand both in total and per hectare. This is explained by two factors. First, wetland quality for high-value wetlands in Svealand is associated with a larger coefficient of variation and larger correlation coefficients. Hence, they contribute less to the certainty equivalent of national wetland quality, and therefore the need for compensation is smaller. Second, a loss of high-quality wetlands in Götaland implies a large cost saving due to the opportunity cost of land foregone, which implies that for costs to remain constant, larger efforts to enhance wetlands can be undertaken elsewhere.

It is noteworthy that the trading ratios between wetland lost and restored are relatively robust to the choice of wetland quality index, given the choice of a national or regional policy scheme. Also, there are modest differences between national and regional policy schemes in trading ratios between wetland area lost and restored in Svealand, which are the only comparable trading ratios.

Finally, creation of habitat is only implemented under regional targets, and in that case mostly in Götaland, where it becomes necessary to apply because restoration is not sufficient to meet the regional target. These trading ratios tend to be relatively high, between 5 and 8 times larger than the trading ratios for restoration, which is explained by the uncertainty attached to this measure.

4.4. Sensitivity analysis

In addition to the above, we made further sensitivity analysis regarding the costs and feasibility of a regional compensation scheme. The sensitivity analysis has similarities to the calculations made for Fig. 3 above, but we now consider reliability levels up to 80%, see Fig. S1 in the Supplementary material. Costs and feasibility were examined for (i) a case where all coefficients of variation were half of the levels applied above, (ii) a case where all coefficients of variation were twice as large as above, and (iii) a case where all correlations were set to zero. This exercise showed that costs as well as feasibility of achieving no-net-loss is strongly affected when the coefficients of variation is doubled. The maximum achievable reliability level under a national scheme when using the red-listed species index then fell from 75% to 65%. In case (i) and (iii), the impact of a higher reliability requirement on costs when using the red-listed species richness index in a national scheme was significantly reduced, and the target could be met at all reliability levels.

5. Discussion and conclusions

The paper applies a novel method to examine ecological compensation, chance constrained programming (CCP), which is useful to derive conditions for trading ratios between different measures to enhance habitats under conditions of uncertainty. The method implies that a safety margin is applied, the size of which is determined by the policy maker's subjectively chosen degree of risk aversion and observed measures of variation and covariation of wetland quality across different wetland types and space. This is useful under conditions where wetland quality is spatially correlated, for example, due to climatic factors. Our work adds to the literature on ecological compensation by analyzing a portfolio problem, where biodiversity values at both conserved sites and sites of compensation is uncertain and spatially correlated. The study also contributes to the economic literature on conservation by using a chance constrained model, which accounts for the role of spatial correlations across habitats, while simultaneously trading ratios across lost habitats versus restoration and construction of habitat can be derived. The use of a chance constrained model such as ours is useful when a

continuous distribution can be assumed, which is relevant for wetland quality in the medium term.

For the purpose of the empirical analysis, we constructed three indices of species richness to measure wetland qualities and compared them with respect to the costs of ecological compensation schemes for agricultural wetlands in Sweden. Results show that costs are more sensitive to the required reliability of target achievement when wetland quality is measured by the red-listed species richness index, compared to the population-weighted species richness index and the species richness index, respectively. The reason for this result is the higher uncertainty about the occurrence of red-listed species in compensation habitats, compared to the exploited habitats in our calculations. We assumed that high-quality wetlands in Götaland were subject to comparatively high exploitation in absolute terms, which is consistent with the actual development. These wetlands provide high expected numbers of red-listed species subject to low uncertainty, which implies that they are valuable from a conservation perspective. Hence, a policy for ecological compensation in Sweden that gives higher weight to red-listed bird species, while simultaneously considering uncertainty, will be more expensive.

There are arguments in the literature for restricting the spatial distance between the exploited site and the site where compensation occurs. These arguments are frequently based on the observation that habitat connectivity is beneficial for biodiversity. Our study adds by recognizing the role that stochasticity plays in such spatial links. We explore the consequences of spatial restrictions on compensation by comparing a scheme that permits compensation on a nationwide basis to one that only permits compensation on regional basis. Based on the theoretical analysis, we note that spatial restrictions imply that compensation schemes become more expensive, as it reduces the possibilities to allocate compensation projects to locations where the ecological effect per euro spent is the highest. However, there is an additional effect: spatial restrictions imply that possible correlation in wetland quality across regions is not accounted for. This implies that aggregate uncertainty could potentially be higher than for a national scheme. A possible consequence is that national targets for wetland quality are not met if targets are set at the regional level. Our empirical analysis confirms the existence of higher costs for regional targets, but it also shows that regional targets result in a higher certainty equivalent wetland quality at the national level. The reason is that regional schemes do not only prevent a cost-efficient allocation of measures but also prevent risk diversification, by ruling out compensations at the national level. Instead, regional schemes necessitate relatively higher efforts to compensate by measures that increase expected wetland quality in regions where this is comparatively expensive.

Earlier studies have shown that the presence of uncertainty is an important factor when deciding on whether compensation payments should target landowner effort or biodiversity outcomes (Gorrdard et al., 2008; Zabel and Roe, 2009; Schilizzi and Latacz-Lohmann, 2016) because this choice determines whether landowners or the environmental manager carries the risk. Our model is useful for policy, as trading ratios can be calculated in terms of land use choices, which are frequently targeted in the agri-environmental policy.

Limitations of the study include the static approach and the fact that the species richness indices used do not fully capture all relevant spatial variations in bird communities. For example, consideration of species abundance is relevant in this context as a full compensation preferably also include the compensation of population losses of species, but reliable time series data on abundance is at present not available at local and across larger spatial scales (see also 3.1. Ecological data). Moreover, ecological compensation across large distances could potentially imply changes in bird species composition, not explicitly dealt with in our study. For example, in our study several wetland species had a predominantly southern distribution (Table S1) which limits the possibilities of fully using national scale for compensation projects. In general, such limits caused by variation in species distributions needs to be

accounted for when large distance biodiversity compensation is considered. Similarly, compensations of impacts on Natura 2000 sites are currently not permitted across biogeographical regions thus limiting the spatial scale of compensations. However, in our case most wetlands except the southern-most ones belong to the same biogeographical region (i.e. boreal). Finally, we do not account for the fact that landowners' willingness to engage in wetland restoration and creation is not solely determined by financial motives but also by interaction and communication within the landowner communities (Aklilu and Elofsson, 2021), which could favor the prospects of ecological compensation in locations where more positive attitudes to the scheme are developed. Given these limitations, a market-like structure of ecological compensation scheme such as examined in this paper is likely to be more useful if applied to valuable, but not unique, habitats and species. The results could therefore be particularly useful if the requirement for compensation would be extended to apply more generally than what is now the case.

Our results suggest interesting avenues for future research, including the development of theoretical and empirical methods for combining ecological compensation based on more general measurements of biodiversity outcomes with instruments that target specific valuable species. This requires empirical methods to identify species that are not well captured by the more general measures. Moreover, empirical

studies that account for the time dynamics of the development of wetland quality at conserved, restored, and constructed habitats should be important for the development of compensation policy schemes.

Declaration of Competing Interest

The authors declare the following financial interests/personal relationships which may be considered as potential competing interests:

Katarina Elofsson reports financial support was provided by The Swedish Environmental Protection Agency.

Data availability

Data will be made available on request.

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Appendix A. Appendix

Table A1

Opportunity cost of agricultural land (SEK/ha), total agricultural area (ha), and standard area of created wetlands (ha).

	Corresponding region	Opportunity cost of agricultural land (SEK/ha)	Agricultural area total (ha)	Standardized area of created wetland (ha)
Blekinge	Götaland	3615	41,607	3
Dalarna	Svealand	563	70,847	4.7
Gotland	Götaland	1384	111,580	17.8
Gävleborg	Norrland	563	72,393	8.7
Halland	Götaland	1789	124,926	4.5
Jämtland	Norrland	330	51,821	5
Jönköping	Götaland	1384	127,112	2.7
Kalmar	Götaland	1384	193,602	3.7
Kronoberg	Götaland	1384	68,044	21.6
Norrbottnen	Norrland	330	35,716	5
Skåne	Götaland	3615	498,640	8.9
Stockholm	Svealand	1699	92,567	2.7
Södermanland	Svealand	1699	142,316	8.9
Uppsala	Svealand	1699	181,425	2.2
Värmland	Svealand	563	113,677	8.5
Västerbotten	Norrland	330	71,497	5
Västernorrland	Norrland	330	50,461	5
Västmanland	Svealand	1699	108,425	5.3
Västra Götaland	Götaland	1789	527,049	2.4
Örebro	Svealand	1699	113,166	2.3
Östergötland	Götaland	1699	243,023	4.3

Sources: SCB (Statistics Sweden) (2019a, 2019b). Standard size wetlands: Own calculations, except for created wetlands which is incurred area from BOA (2018).

Table A2a

Correlation between aggregate wetland habitat quality across wetland type and region, species richness index.

	Götal., hi. qual.	Sveal., hi. qual.	Götal., med. qual.	Sveal., med. qual.	Norrl., med. qual.
Götal., hi. qual.	1	0.841	0.394	0.857	0.763
Sveal., hi. qual.	0.841	1	0.736	0.867	0.496
Götal., med. qual.	0.394	0.736	1	0.650	0.772
Sveal., med. qual.	0.857	0.867	0.650	1	0.931
Norrl., med. qual.	0.763	0.496	0.772	0.931	1

Note: The correlation is calculated as $Cov(b_{jd}, b_{k\neq j, e\neq d}) / [\sqrt{Var(b_{jd})} \sqrt{Var(b_{k\neq j, e\neq d})}]$, using data from 2005 to 2014.

Table A2b

Correlation between aggregate wetland habitat quality across wetland type and region, population-weighted species richness index.

	Götal., hi. qual.	Sveal., hi. qual.	Götal., med. qual.	Sveal., med. qual.	Norrl., med. qual.
Götal., hi. qual.	1	0.850	0.850	0.850	0.850
Sveal., hi. qual.	0.850	1	0.703	0.764	0.817
Götal., med. Qual.	0.420	0.703	1	0.632	0.689
Sveal., med. Qual.	0.637	0.764	0.632	1	0.843
Norrl., med. Qual.	0.719	0.817	0.689	0.843	1

Note: The correlation is calculated as $Cov(b_{jd}, b_{k\neq j, e\neq d}) / [\sqrt{Var(b_{jd})} \sqrt{Var(b_{k\neq j, e\neq d})}]$, using data from 2005 to 2014.**Table A2c**

Correlation between aggregate wetland habitat quality across wetland type and region, red-listed species richness index.

	Götal., hi. qual.	Sveal., hi. qual.	Götal., med. qual.	Sveal., med. qual.	Norrl., med. qual.
Götal., hi. qual.	1	0.474	0.611	0.807	0.696
Sveal., hi. qual.	0.474	1	0.709	0.843	0.855
Götal., med. Qual.	0.611	0.709	1	0.634	0.508
Sveal., med. Qual.	0.807	0.743	0.634	1	0.828
Norrl., med. Qual.	0.696	0.855	0.508	0.828	1

Note: The correlation is calculated as $Cov(b_{jd}, b_{k\neq j, e\neq d}) / [\sqrt{Var(b_{jd})} \sqrt{Var(b_{k\neq j, e\neq d})}]$, using data from 2005 to 2014.**Appendix B. Supplementary data**Supplementary data to this article can be found online at <https://doi.org/10.1016/j.ecolecon.2022.107672>.**References**

- Aklilu, A.Z., Elofsson, K., 2021. Wetland investment support schemes: adoption and spatial interactions. *Land Econ.* 98 (2), 292–313.
- Bateman, I.J., Day, B.H., Georgiou, S., Lake, I., 2006. The aggregation of environmental benefit values: welfare measures, distance decay and total WTP. *Ecol. Econ.* 60 (2), 450–460.
- Blicharska, M., Hedblom, M., Josefsson, J., Widenfalk, O., Ranius, T., Öckinger, E., Widenfalk, L.A., 2022. Operationalisation of ecological compensation—obstacles and ways forward. *J. Environ. Manag.* 304, 114277.
- BOA, 2018. Våtmarker och dammar – en uppföljning inom landsbygdsprogrammet. Board of Agriculture, Jönköping. As available 2019-09-13 at: <http://www.jordbruksverket.se/download/18.710c5e85168e83df266748ce/1550219818088/Uppf%C3%B6ljning%20st%C3%B6d%20till%20v%C3%A5tmarker%20och%20dammar.pdf>.
- Boisvert, V., 2015. Conservation banking mechanisms and the economization of nature: an institutional analysis. *Ecosyst. Serv.* 15, 134–142.
- Briggs, B.D., Hill, D.A., Gillespie, R., 2009. Habitat banking—how it could work in the UK. *J. Nat. Conserv.* 17 (2), 112–122.
- Charnes, A., Cooper, W.W., 1959. Chance-constrained programming. *Manag. Sci.* 6 (1), 73–79.
- Charnes, A., Cooper, W.W., 1963. Deterministic equivalents for optimizing and satisfying under chance constraints. *Oper. Res.* 11 (1), 18–39.
- Drechsler, M., Wätzold, F., 2009. Applying tradable permits to biodiversity conservation: effects of space-dependent conservation benefits and cost heterogeneity on habitat allocation. *Ecol. Econ.* 68, 1083–1092.
- Fahrig, L., 2020. Why do several small patches hold more species than few large patches? *Glob. Ecol. Biogeogr.* 29 (4), 615–628.
- Flyckt, L., 2010. Reningsresultat, drifterfarenheter och kostnadseffektivitet i svenska våtmarker för spillvattenrening. Linköping University. Accessed on: 2019-04-09, Available from: https://wrs.se/wp-content/uploads/2014/03/Exjobb_LindaFlyckt110126_svenska-vatmarker-for-spillvattenrening.pdf.
- GAMS Development Corporation, 2019. General Algebraic Modeling System (GAMS) Release 27.1.0. Fairfax, VA, USA.
- Gorddard, R.J., Whitten, S.M., Reeson, A., 2008. When Should Biodiversity Tenders contract on Outcomes? (No. 419-2016-26579).
- Greene, W.H., 1993. *Econometric Analysis*. Macmillan, New York.
- Gren, M., Destouni, G., Tempono, R., 2002. Cost effective policies for alternative distributions of stochastic water pollution. *J. Environ. Manag.* 66 (2), 145–157.
- Gren, M., Baxter, P., Mikusinski, G., Possingham, H., 2014. Cost-effective biodiversity restoration with uncertain growth in forest habitat quality. *J. For. Econ.* 20 (1), 77–92.
- Hansen, A.J., DeFries, R.S., Turner, W., 2012. Land use change and biodiversity. In: *Land Change Science*. Springer, Dordrecht, pp. 277–299.
- Hartig, F., Drechsler, M., 2009. Smart spatial incentives for market-based conservation. *Biol. Conserv.* 142, 779–788.
- Hiron, M., Pärt, T., Siriwardena, G.M., Whittingham, M.J., 2018. Species contributions to single biodiversity values under-estimate whole community contribution to a wider range of values to society. *Sci. Rep.* 8 (1), 1–7.
- Hoekstra, J., 2012. Improving biodiversity conservation through modern portfolio theory. *Proc. Natl. Acad. Sci.* 109 (17), 6360–6361.
- Hogg, R.V., Craig, A.T., 1995. *Introduction to mathematical statistics*. (5 edition). Englewood Hills, New Jersey.
- IPBES, 2019. In: Díaz, S., Settele, J., Brondizio, E.S., Ngo, H.T., Guèze, M., Agard, J., Arnett, A., Balvanera, P., Brauman, K.A., Butchart, S.H.M., Chan, K.M.A., Garibaldi, L.A., Ichii, K., Liu, J., Subramanian, S.M., Midgley, G.F., Miloslavich, P., Molnár, Z., Obura, D., Pfaff, A., Polasky, S., Purvis, A., Razaque, J., Reyers, B., Chowdhury, R. Roy, Shin, Y.J., Visseren-Hamakers, I.J., Willis, K.J., Zayas, C.N. (Eds.), Summary for Policymakers of the Global Assessment Report on Biodiversity and Ecosystem Services of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services. IPBES secretariat, Bonn, Germany.
- Josefsson, J., Widenfalk, L.A., Blicharska, M., Hedblom, M., Pärt, T., Ranius, T., Öckinger, E., 2021. Compensating for lost nature values through biodiversity offsetting—where is the evidence? *Biol. Conserv.* 257, 109117.
- Kačergytė, I., 2021. Wetland Creation and Restoration for Biodiversity: Outcomes of Conservation Initiatives to Benefit Birds, Amphibians and Occasionally Fish [PhD thesis] Acta Universitatis Agriculturae Sueciae, 1652–6880. Swedish University of Agricultural Sciences, Uppsala.
- Kačergytė, I., Arlt, D., Berg, Å., Zmihorski, M., Knape, J., Rosin, Z.M., Pärt, T., 2021. Evaluating created wetlands for bird diversity and reproductive success. *Biol. Conserv.* 257, 109084.
- Kangas, J., Ollikainen, M., 2019. Economic insights in ecological compensations: market analysis with an empirical application to the Finnish economy. *Ecol. Econ.* 159, 54–67.
- Mallory, M.L., Ando, A.W., 2014. Implementing efficient conservation portfolio design. *Resour. Energy Econ.* 38, 1–18.
- Markowitz, H., 1952. Portfolio analysis. *J. Financ.* 8, 77–91.
- Markowitz, H., 1959. Portfolio Selection. Investment Under Uncertainty.
- Millennium Ecosystem Assessment, 2005. Ecosystems and human well-being. In: *Biodiversity Synthesis*. <http://www.millenniumassessment.org/documents/docum ent.354.aspx.pdf>.
- Moilanen, A., van Teuffelen, A.J.A., Ben-Haim, Y., Ferrier, S., 2009. How much compensation is enough? A framework for incorporating uncertainty and time discounting when calculating offset ratios for impacted habitat. *Restor. Ecol.* 17, 470–478.
- Österling, L., Kindt, T., 2007. Projektkatalog: Våtmarksprojektet i Laholms kommun. Accessed on: 2019-04-09, Available from: https://www.laholm.se/globalassets/uplod/miljo-och-byggnadsnamnd/miljokontoret/projekt/vatmarksprojekt_projektat alogr.pdf.
- Ottosson, U., Ottval, R., Elmberg, J., Green, M., Gustafsson, R., Haas, F., Holmqvist, N., Lindström, Å., Nilsson, L., Svensson, M., Svensson, S., Tjernberg, M., 2012. Fåglarna i Sverige – antal och förekomst. SOF, Halmstad.
- Paris, Q., Easter, C.D., 1985. A programming model with stochastic technology and prices: the case of Australian agriculture. *American Journal of Agricultural Economics* 67 (1), 120–129.
- Polasky, S., Nelson, E., Lonsdorf, E., Fackler, E., Starfield, A., 2005. Conserving species in a working landscape: land use with biological and economic objectives. *Ecol. Appl.* 15, 1387–1401.
- Rodrigues, A.S., Pilgrim, J.D., Lamoreux, J.F., Hoffmann, M., Brooks, T.M., 2006. The value of the IUCN red list for conservation. *Trends Ecol. Evol.* 21 (2), 71–76.
- Ruete, A., Pärt, T., Berg, Å., Knape, J., 2017. Exploiting opportunistic observations to estimate changes in seasonal site use: an example with wetland birds. *Ecol. Evol.* 7 (15), 5632–5644.

- SCB (Statistics Sweden), 2019a. Agricultural Land Rents 2018. JO 39 SM 1901.
- SCB (Statistics Sweden), 2019b. Agricultural Land. Database.
- Schilizzi, S., Latacz-Lohmann, U., 2016. Incentivizing and tendering conservation contracts: the trade-off between participation and effort provision. *Land Econ.* 92 (2), 273–291.
- SEPA, 2016. Ecological compensation. In: Handbook 2016:1. Swedish Environmental Protection Agency, Stockholm.
- Seto, K.C., Güneralp, B., Hutyra, L.R., 2012. Global forecasts of urban expansion to 2030 and direct impacts on biodiversity and carbon pools. *Proc. Natl. Acad. Sci.* 109 (40), 16083–16088.
- Shah, P., Ando, A.W., 2015. Downside versus symmetric measures of uncertainty in natural resource portfolio design to manage climate change uncertainty. *Land Econ.* 91 (4), 664–687.
- Shah, P., Mallory, M.L., Ando, A.W., Guntenspergen, G.R., 2017. Fine-resolution conservation planning with limited climate-change information. *Conserv. Biol.* 31 (2), 278–289.
- Swedish Species Observation System, 2015. Rödlistade arter i Sverige 2015. ArtDatabanken SLU, Uppsala.
- Takayama, A., 1993. *Analytical Methods in Economics*. University of Michigan Press.
- van Teeffelen, A.J., Opdam, P., Wätzold, F., Hartig, F., Johst, K., Drechsler, M., Vos, C.C., Wissel, S., Quétier, F., 2014. Ecological and economic conditions and associated institutional challenges for conservation banking in dynamic landscapes. *Landscape and Urban Planning* 130, 64–72.
- 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. *Biol. Conserv.* 123 (3), 327–338.
- Wissel, S., Wätzold, F., 2010. A conceptual analysis of the application of tradable permits to biodiversity conservation. *Conserv. Biol.* 24 (2), 404–411.
- Zabel, A., Roe, B., 2009. Optimal design of pro-conservation incentives. *Ecol. Econ.* 69 (1), 126–134.