Cost-effective Measures to Enhance Biodiversity on the Swedish Agricultural Plains

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Abstract

The general objective of this study is to identify measures that promote biodiversity in the plain districts of southern Sweden in a cost-effective manner. To achieve this goal we rank different conservation measures according to cost-effectiveness. We also account for uncertainty about the impact of measures on biodiversity. The effect of a measure is then evaluated as the certainty equivalent of the impact on biodiversity, which is determined by the variability of the impact, and the policy makers' risk aversion. The study focuses on improvements in bird species richness and abundance through changes in agricultural practices in the plain districts in Svealand (SS) and Southern Götaland (GSS). Differences in cost-effectiveness between measures and agricultural districts as well as between conventional and organic farming systems are examined. Results show that a reduction of the size of fields would be the cheapest measure to improve both abundance and richness, followed by increased areas of pasture. Introduction of grassy field margins could also improve bird abundance at low cost. Reductions in yield, conversion to organic farming and increased landscape heterogeneity are all comparatively expensive measures. The difference in cost-effectiveness between low- and high-cost measures increases when uncertainty is accounted for as the relative uncertainty about impact of high-cost measures is larger. Conversion to organic farming and increased landscape heterogeneity are more expensive in GSS compared to SS, which is explained by the higher productivity of arable land in the GSS region. Field size reductions are even more cost-effective when applied at organic farms, which is explained by the comparatively smaller scale economies in cereal farming on organic farms compared to conventional farms. Important limitations to the study are the static perspective, exclusion of interaction effects between measures and exclusion of transaction costs, all explained by lack of data in the literature.

Keywords: cost-effectiveness, biodiversity, bird richness, bird abundance, agriculture

Introduction

Intensified cultivation of arable land is a major threat to biodiversity. Increased use of agrochemicals and modern machinery has allowed for larger and more intensively managed fields and the removal of farmland characteristics such as field margins, ditches, and non-arable outcrops that provide important habitats for various species (Stoate et al., 2001; Stoate et al., 2009). The two-folded process of agricultural intensification in some regions, and abandonment in others, contributes to the simplification of landscapes, reducing the number of habitats, and causing species richness and abundance across various trophic taxa to decline (Fuller et al., 2005; Wretenberg et al. 2006; Winqvist et al., 2012;). Agricultural land constitutes the major land use in Europe, and consequently, a high percentage of Europe's biodiversity depends on agricultural areas for its persistence (Hole et al., 2005). In Sweden, two thirds of red-listed herbs, one third of red-listed birds, half of the insects, and three quarters of the reptiles, are dependent on agricultural environments (SSNC, 2011).

Through the introduction of specific agri-environmental measures (AEMs) in the 1980s, policy makers attempt to maintain and enhance natural resources, biodiversity, and landscape values. In 2002, measures to promote biodiversity were implemented on as much as 25 per cent of the agricultural land within the EU, and a considerable amount of resources is scheduled for AEMs from both the EU and domestic sources (Primdahl et al., 2010; Höjgård and Rabinowicz, 2011). In the Swedish Rural Development Program 2007-2013, close to 80 per cent of the Axis 2 budget is allocated to AEMs and non-productive investments, see table 1 (Höjgård and Rabinowicz, 2011). The political aim behind compensations for AEMs is to balance the provision of public goods, such as biodiversity, without compromising the production of private goods in the agricultural sector. Consequently, the implementation of current AEMs within the EU is based on voluntary schemes, where farmers are provided with economic incentives for adoption of biodiversity conservation measures.

Measure	**Total budget (m €)**	**Share of budget (%)**
Natural handicap payments to farmers in mountain areas, etc.	562	21
Agri-environmental payments and support for non-productive investments	2106	78
Support for non-productive investments in forestry	34	1
Axis 2 Total	*2702*	*100*

Table 1. Allocation of the Swedish budget under Axis 2 for 2006-2013 (Höjgård and Rabinowicz; 2011).

The choice of measures for which compensation is paid is largely based on qualitative information about the impact on biodiversity, while the magnitude of the impact is, typically, not well known to policy-makers (Primdahl et al., 2010). Each member state determines the level of compensation paid to farmers and the payments are not linked directly to environmental outcomes, but based on profits forgone and operation and management costs (Wätzold and Schwerdtner, 2005; Primdahl et al., 2010).

Despite the substantial funding for improved biodiversity in the agricultural landscape, there are few studies that evaluate the achieved impact of conservation efforts. This is partly due to difficulties in assessing biodiversity impacts, and the fact that several measures target multiple objectives, e.g. both cultural and environmental (Ministry of the Environment, 2010; Smith and Ahnström, 2010). Consequently, the implementation of different measures and their spatial distribution are likely to result in smaller improvements in biodiversity than would be the case if resources were optimally allocated (EPA, 2010; Board of Agriculture, 2008).

Due to the above, current policy is not likely to be cost-effective, i.e. biodiversity objectives of the Swedish Parliament and in the Rural Development Program are not attained at least cost. Yet cost-effectiveness is, in general, a criterion of high importance to policy-makers, if they wish to allocate a scarce environmental policy budget to different activities such that the highest possible environmental improvement is achieved. In the case of biodiversity, cost-effectiveness requires information on the magnitude of the impact of different measures on biodiversity, as well as the cost of the associated measure. Adding to the difficulties to determine cost-effectiveness of policies to promote biodiversity in the agricultural sector is the uncertainty about the impact of a given measure in a certain location, as the impact is dependent on locally specific conditions. Further, uncertainty about the impact also varies between measures. Consequently, if policy makers are concerned about risk, it is necessary for them to evaluate not only the expected impact in relation to the costs incurred but also to judge the importance of the risk inherent in different measures.

Few empirical studies couple economic and ecological aspects of measures with in order to determine cost-effectiveness. Those available typically model situations where the aim is to protect a single species, see e.g. Johst et al. (2002), to model biodiversity as a function of a single activity, e.g. fertilization intensity or mowing regimes (Groot et al., 2007; Johst et al., 2002) or simulate the impact on species from different policy scenarios (Oglethorpe and Sanderson, 1999). Single species indicators are also used to evaluate the effectiveness of measures to prevent a species from extinction (Drechsler et al, 2007b). Most studies are based on the assumption of a limited public budget, but differ in the choice of objective function, which can be expressed as, e.g., maximization of a biodiversity indicator, minimization of the risk of extinction or maximization of species perseverance (Nicholson and Possingham, 2006; Nicholson et al., 2006).

Several studies calculate the compensation necessary to make farmers carry out certain management practices (Drechsler et al., 2007a; Oglethorpe and Sandersson, 1999). Such compensation is needed to cover the costs for enhanced biodiversity that arise to farmers, e.g. due to profits foregone, management activities and time consumed in decision-making. Differences in costs between farmers arise due to variations in soil quality, farmer experience, equipment availability, and opportunity costs for labour and land (Drechsler et al., 2001; Wätzold and Dreschler, 2004). In contrast with the widespread use of spatially uniform measures and payments in actual policy, research shows that compensation should be differentiated over time and in space if cost-effectiveness is to be achieved (Dreschler and Wätzold, 2001, 2007; Johst et al., 2002; Wätzold and Dreschler, 2004). Further, larger farms are more likely to participate in biodiversity preservation programs compared to small farms and this difference is likely to be explained by economic, rather than ecological, factors (Larsén 2006; Morris and Potter, 1995; Wilson, 1997). Given the widespread use of uniform compensation schemes, and cost variations that are related to farm characteristics and farm location, the current spatial distribution of a given type of measure is thus likely to be largely determined by economic, rather than ecological factors.

Not only variations in cost, but also the functional relationship between implemented measures and resulting biodiversity change is important to policy choice. This functional relationship can, e.g., involve thresholds, which affects the cost-effective allocations of efforts over space and time (Wu and Bogess, 1999). As noted above, the relationship between efforts and associated biodiversity effects are location-specific, wherefore results from a given location cannot automatically be generalized (Johst et al., 2002).

The purpose of this study is to analyze the cost-effectiveness of different measures to improve biodiversity in the agricultural landscape, while taking into account the inherent uncertainty about the impact of different measures on biodiversity. In order to achieve this, we first review determinants of biodiversity on arable land in order to identify relevant measures, and the mechanisms behind their impact on biodiversity. Second, we calculate the costs and the expected impact of measures, and compare the policy conclusions that would be drawn if policy decisions would be based solely on ecological or economic information. This is relevant as measures currently subsidized are included in policies, and compensation is paid, based on highly limited information. Third, we aim at ranking measures based on cost-effectiveness, acknowledging that policy-makers' attitude to risk can have an impact on the ranking. Fourth, we investigate whether there are systematic differences in cost-effectiveness between regions and farm types. Finally, we acknowledge that available estimates of ecological impacts are uncertain, given the limited quantitative research in the area. We therefore carry out sensitivity analysis in order to find out whether results are robust with respect to uncertainty about the expectation and variance of the ecological impact.

In order to achieve the above, we investigate the impact of and cost for improved biodiversity, using bird abundance and bird richness as indices of biodiversity. Functional relationships between agricultural practices and the associated impact on biodiversity, including uncertainty, are obtained from the ecological literature. Costs for changes in the same agricultural practices are calculated as the private costs to farmers. Uncertainty is accounted for through the calculation of a certainty equivalent of each measure, which takes into account variability of the impact of the measure, as well as the policy maker's subjectively chosen reliability level. The study is applied to agricultural measures on cropland in the Plain Districts in Svealand (SS) and Southern Götaland (GSS). Differences in cost-effectiveness between measures and agricultural districts as well as between conventional and organic farming systems are examined. The study differs from earlier studies mainly through the investigation of the role of uncertainty for the choice of policy measure, and through the application to the Swedish plains. Results can contribute to the development of more cost-effective policies for enhanced biodiversity in the Swedish agricultural plains. Important limitations are the static perspective and the exclusion of transaction costs.

The study is organized as follows: first, we briefly review determinants of biodiversity on arable land. Thereafter, we present method and data, and describe the study area. This is followed by a presentation of the results. The paper ends with a discussion.

Determinants of biodiversity on agricultural land

Similar to other countries in Northern Europe, Sweden is experiencing a simultaneous field abandonment and (although less dramatic) agricultural intensification depending on geographical region (Wretenberg et al. 2007). In forest-dominated regions where cultivated grasslands dominate agriculture, an extensification process with concurrent abandonment of active farms is occuring. One potential explanation for the recent development in these regions is that stricter criteria have been introduced for supportive payments to grassland (Board of Agriculture, 2010). In plain regions, dominated by intensive cereal farming, the proportion of farms with animal husbandry has decreased, and the sizes of individual farm holdings have increased, causing removal of non-crop habitats (Ihrse, 1995). One key driver of this development is increasing economies of scale, benefiting the development of large, intensively managed farms (Helmfrid and Björklund, 2010). This two-folded process of abandonment and intensification is impacting biodiversity negatively.

The importance of landscape heterogeneity in supporting biodiversity is well established. The Swedish Environmental Quality Objective “A Varied Agricultural Landscape” emphasizes the importance of a heterogeneous and varied agricultural landscape, while simultaneously supporting an economically rational and competitive agricultural production. Many studies and reviews have found that heterogeneous landscapes support a higher biodiversity across a range of taxa (e.g. Tscharntke et al. 2005, Hole et al. 2005, Benton et al. 2003). Heterogeneous landscapes maintain this diversity by providing a high diversity of habitats and thus increasing the number of possible niches, an increased number of potential food supplies, and a diversity of refuge sites (Benton et al., 2003). The effects of landscape heterogeneity on local populations differ between taxa, as different species have different requirements and operate on different spatial and temporal scales due to differing dispersal ability (Jonason et al., 2012; Concepción and Díaz, 2011; Tscharntke et al., 2005, Söderström and Pärt, 2000). Swedish studies confirm the role of landscape heterogeneity for biodiversity (Rundlöf and Smith, 2006; Weibull et al., 2000; Weibull and Östman, 2003; Smith et al., 2010). Furthermore, the relationship between landscape and biodiversity has implications for important ecosystem functions (Altieri, 1999). For example, a study by Holzschuh et al. (2008) found that a high semi-natural land cover increases the abundance of pollinators. Landscape heterogeneity can also influence crop pest control, being highest in complex landscapes and declining with increasing landscape homogeneity (Winqvist et al. 2012).

Recent studies have found that landscape composition not only moderate biodiversity, but also the ecological impacts of AEMs. Changes in land-use intensity have greater impact on biodiversity when undertaken on extensively farmed land than on intensively used farmland, see Figure 1 (Kleijn and Sutherland, 2003; Kleijn et al. 2011). Along the same line, effects of reduced intensity are stronger in simple landscapes compared to more complex landscapes (Tscharntke et al., 2005; Batáry et al. 2011). This hypothesis, recently dubbed the intermediate landscape-complexity hypothesis (Tscharntke et al., 2012), states that the weaker effects of AEMs in complex landscapes result from a continuous re-colonisation from neighbouring fields, which compensates for the negative effects of intensified cultivation, while cleared landscapes do not have source populations to sustain re-colonisation, thus eradicating any effects of reductions in farming intensity (Concepción et al., 2007).

Furthermore, the importance of landscape heterogeneity for the biodiversity impact of measures often differs between species (Bátary et al., 2011). For example, landscape characteristics can interact with life-history traits, i.e. reproduction and survival patterns of the species, with regard to the effects of measures on farmland bird abundance. Small species benefit relatively more from local on-field measures whereas larger birds are more responsive to changes in the wider landscape context (Concepción and Díaz, 2011, Fisher and Owens, 2004). Also, it is worth noting that environmental changes do not always result in immediate or even swift changes in biodiversity, but rather occur over longer time periods, even decades (Jonason et al., 2011).

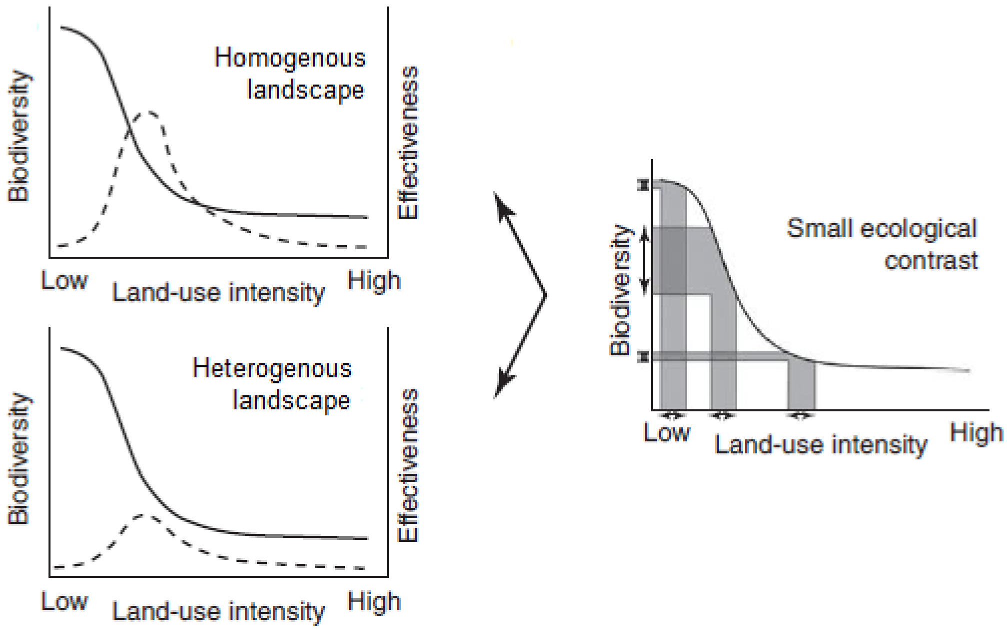

Figure 1. Landscape Structure and Biodiversity Effects (adapted from Kleijn et al., 2006). Dashed lines describe the effectiveness of conservation measures, expressed as reduced land-use intensity, and the solid lines represent biodiversity.

Studies on the effects of conversion to organic farming on local biodiversity in relation to landscape heterogeneity in Sweden give support to the intermediate landscape-complexity hypothesis. In the homogenous agricultural landscape of GSS, organic farming increases abundance and richness of representative trophic taxa (Rundlöf and Smith, 2006; Rundlöf et al., 2008; Smith et al., 2010), whereas in the more heterogeneous landscapes of SS, the effect is smaller or lacking (Weibull et al., 2000; Weibull and Östman, 2003; Belfrage et al., 2005). The effect of certain AEMs, such as the introduction of low-intensity land-use on arable land could also depend on the relative "commonness" or abundance of the measure in the landscape (Wretenberg et al., 2010), suggesting that the impact is non-linear. This is referred to as a "rare habitat effect", meaning that measures that are implemented in homogeneous landscapes and alter the common landscape characteristics can have a higher effect than measures introduced in heterogeneous landscapes.

Studies examining existing AEMs, suggest that their impact on biodiversity could be improved, e.g. by deploying a higher proportion of measures in complex landscapes, and through clustering of AEMs (Whittingham, 2011; Kleijn et al., 2011). In a meta-analysis by Batáry et al. (2011), species richness and abundance increased in such diverse AEMs such as reductions in agrochemical input, soil cultivation, mowing frequency and cattle density, as well as enhancement of organic farming and field margin strip cultivation, many of which are eligible for subsidies within in one or more European countries.

Method

A major purpose of the study is to rank AEMs according to cost-effectiveness, taking into account uncertainty of the impact on biodiversity. In this section we therefore first explain the method used to rank measures. Thereafter, the calculation of the different sub-components, necessary for the calculation of the ranking, is explained in further detail. These subcomponents are the biodiversity indicators, the biodiversity effects and the costs of different measures. Finally, characteristics of the chosen study area are outlined.

Cost-effectiveness ratio under uncertainty

In order to account for uncertainty, we calculate the certainty equivalent of the impact of the measure. The certainty equivalent is calculated as the expected effect minus an uncertainty discount. This approach is widely used in industrial economics and water economics to evaluate outcomes when there is uncertainty about technology. In the case that we study here, the uncertainty about the effect of measures can be seen as uncertainty about nature's technology, i.e. about the functional relationship between an input, an AEM, and the corresponding output, the biodiversity.

The methodology is based on the presumption that a policy-maker wants to achieve a given output at, at least, a given probability (see e.g. Taha 1976; Charnes and Cooper, 1963). The biodiversity effect is then evaluated in deterministic terms as the expected value minus a risk discount, where the latter is determined by the subjectively chosen probability, the probability distribution of the impact and the variance of the impact. The certainty equivalent, Q_c, for a measure that improves biodiversity can then calculated as:

$$Q_c = E(Q) - z_\alpha \sigma$$

where $E(Q)$ is the expected effect of the measure, z_a is a function of the desired level of confidence α for Q_c, and σ is the variance (see e.g. Kim and McCarl, 2009). In statistical terms, we can then calculate a lower limit of the quantity of biodiversity generated for a desired confidence level from this equation. Such a formula, in a one tailed context, reduces the amount of the uncertain quantity until one reaches a level that exhibits a particular probability level α that Q_c or more will be produced. Frequently, a normal distribution of the random variable is assumed where for example a z_a value of 1.64 implies α = 95%. One can also convert this formula in terms of the coefficient of variation (CV) where the formula for Q_c becomes:

$$Q_c = E(Q) - z_\alpha CV \cdot E(Q)$$

which is the form we will use and where the uncertainty discount factor would be $z_\alpha CV$. In the following calculations we assume a normal distribution of the random variable and consider confidence levels ranging from 50% to 95%. A 50% confidence level is one where the policy maker is risk neutral, i.e. does not care about uncertainty about the biodiversity impact implying that only the expected impact matters.

A higher confidence level implies that the policy maker values the measures lower if it is uncertain, and the reduction in the value is larger if the measure is more uncertain.

If we want to evaluate the cost-effectiveness of the measure, the certainty equivalent must be related to the cost of the measure. Consequently, it is necessary to collect information not only on the expected impact and the variation of the impact but also on the costs for implementing the measure. The cost-effectiveness of the measure is then evaluated in terms of a cost-effectiveness ratio Q_c/AC where AC is the unit cost of the measure.

Costs and effects are, when possible, calculated for both organic and conventional farms. Exceptions are increased landscape heterogeneity and conversion from conventional to organic farming, where such differentiation is not relevant.

The remainder of this chapter first describes the study area, and then biodiversity data relevant to the study area are presented. This is followed by a description of the calculation of the costs and effects of the included measures.

Biodiversity indicators

Among ecologists, taxa at different trophic levels (e.g. vascular plants, arthropods and birds) are frequently used as general indicators of how biodiversity responds to environmental changes in agricultural landscapes and several studies have found a positive correlation between the abundance of birds and the abundance of lower trophic taxa such as butterflies and herbaceous plants (Billeter, et al., 2008; Belfrage et al, 2005). Therefore, measures of bird diversity are often used as indicators of general biodiversity (Wretenberg et al., 2010). We follow this approach by using bird abundance, i.e. the number of bird individuals per hectare, and species richness, i.e. the number of bird species per hectare, as measures of biodiversity.

The present biodiversity, i.e. the baseline biodiversity, is calculated for each crop and farm type in the two regions. Bird biodiversity data from inventories conducted in a cooperative network of birdwatchers and farmers promoting birdlife on farms in Sweden are used for these calculations (Eggers and Engström, 2007)[1]. Mean values were calculated for ecologically and conventionally cultivated fields as well as for average fields[2] for 29 common agricultural bird species (see Appendix) and for different crop types. Observations in different crop types are weighted according to their share in total agricultural area in each region, respectively, whereby average, baseline biodiversity is obtained. Data on bird abundance and species richness is missing for many organically cultivated crops, wherefore the corresponding figures have been calculated assuming that the variation between crops is the same for both conventional and organic farms. Table 2 and 3 show data on observed bird abundance and richness for different crop types, as well as the calculated crop area-weighted abundance and richness baselines for conventional and organic farming and for total agricultural land.

[1] Collection of data and preparation of database is co-financed by Formas, the Swedish Environmental Protection Agency and FOMA.
[2] I.e. with representative shares of conventional/organic farming, respectively.

	GSS All arable land		GSS Conventional		GSS Organic	
	Abundance # of individuals	**Richness # species**	**Abundance # of individuals**	**Richness # species**	**Abundance # of individuals**	**Richness # species**
Autumn wheat	0,62	0,41	0,67	0,43	NA	NA
Spring wheat	0,57	0,44	0,57	0,44	NA	NA
Rye	0,47	0,36	0,49	0,38	NA	NA
Autumn Barley	0,46	0,29	0,51	0,31	NA	NA
Spring Barley	0,79	0,50	0,92	0,58	0,67	0,67
Oat	1,47	1,09	1,47	1,09	NA	NA
Triticale	1,44	0,78	1,44	0,78	NA	NA
Mixed seed	1,10	0,44	1,10	0,44	NA	NA
Pasture	0,95	0,72	1,08	0,84	0,57	0,76
Autumn Rapeseed	0,52	0,34	0,64	0,41	NA	NA
Spring Rapeseed	0,37	0,37	0,37	0,37	NA	NA
Fallow	3,01	2,63	3,21	2,81	NA	NA
Pasture	1,42	1,03	1,44	1,23	0,66	1,34
Average weighted abundance/richness[3]	0,73	0,48	0,80	0,52	1,11	0,96

Table 2. GSS total, conventional and organic baseline for biodiversity

Biodiversity effects

The effect of a measure on biodiversity is, in this study, calculated as the absolute increase in bird abundance or richness in the area where the measure is applied, compared to the level of the biodiversity measures if the area was managed according to business-as-usual. Information about the biodiversity effect and the standard deviation thereof is obtained from the ecological literature and from own analysis of farm level data from the study regions.

Several of the biodiversity effects are obtained from Geiger et al. (2010), where the impact of various agricultural practices and landscape characteristics on abundance and richness of farmland birds is estimated using cross-section data from seven study areas in six European countries, including Sweden. The variation in sites studied implies that results are likely to be more robust compared to studies conducted only at a single site. Moreover, the study differs from most other research on the subject by inclusion of quantitative estimates, including standard deviations, of the effect of measures and landscape features on biodiversity in terms of bird abundance and richness. Given that Geiger et al. (2010) report only the percentage improvement in biodiversity, but do not report baseline biodiversity, the biodiversity effect, in absolute terms, is here calculated using the baseline biodiversity data reported in tables 2 and 3. Details about the conversion of estimated coefficients and standard deviation from results reported in

[2] Sum of the number of species/birds per hectare crop multiplied by the respective proportional crop share of total crop areal.

	SS All arable land		SS Conventional		SS Organic	
	Abundance # of individuals	**Richness # species**	**Abundance # of individuals**	**Richness # species**	**Abundance # of individuals**	**Richness # species**
Autumn wheat	0,83	0,59	0,88	0,65	0,84	0,55
Spring wheat	0,63	0,49	0,55	0,29	0,65	0,56
Rye	0,60	0,48	0,53	0,43	0,53	0,44
Autumn Barley	NA	NA	NA	NA	NA	NA
Spring Barley	0,59	0,42	0,59	0,42	0,41	0,31
Oat	0,73	0,52	0,67	0,46	0,74	0,56
Triticale	1,06	0,50	1,11	0,71	1,38	0,39
Mixed seed	0,44	0,35	0,00	0,00	0,46	0,36
Pasture	1,62	1,23	1,71	1,23	1,49	1,16
Autumn Rapeseed	0,87	0,65	0,81	0,46	0,56	0,56
Spring Rapeseed	0,98	0,63	0,98	0,63	NA	NA
Fallow	1,53	1,10	2,33	1,69	0,62	0,49
Pasture	1,15	0,84	1,35	1,02	1,03	0,73
Average weighted abundance/richness[4]	0,71	0,50	0,73	0,53	0,69	0,44

Table 3. SS total, conventional and organic baseline for biodiversity.

Geiger et al. (2010) into percentage and absolute figures for the purpose of this report are provided in the Appendix.

Notably, there are some scientific studies applied directly to GSS and SS, where the relationship between measures and biodiversity is investigated, but these studies only report qualitative results that ascertain that the measure has an impact, but not the magnitude of that impact (Rundlöf and Smith, 2006; Wretenberg et al., 2007; Smith et al., 2010). However, it can be observed that these studies reach similar qualitative conclusions as Geiger et al. (2010), as regards the statistical significance of measures.

Although the more qualitatively oriented ecological literature argues that there are cross-effects between different measures and landscape characteristics, there are, to our knowledge, no studies that report the magnitude of these cross-effects. Therefore, in this study, we assume separability between measures with regard to their impact on biodiversity. Also, given the lack of empirical, quantitative evidence on the role of scale for the impact on biodiversity, we assume a linear relationship between measures and the corresponding biodiversity impact.

Cost Calculations

Costs arise due to changes in outputs from and/or inputs in production that result from the use of a particular measure. The cost of the measure is then the net loss of profit from implementing the measure, compared to *status quo*. Prices on agricultural inputs and outputs are assumed to be exogenous, which is plausible in the regional context analyzed in combination with the mainly national and international markets for agricultural products. In addition, the total area of arable land is assumed fixed in the short run in each region.

The costs of changes in agricultural crop production are, in the following, calculated as the short-run private cost of farmers' changes in production practices. For changes in crop choice and land use, this cost is calculated as the opportunity cost. The opportunity cost is then defined as the change in the per hectare contribution margin compared to cultivation of the current, typical crop mix on fields of 200 ha in the region in question. The typical crop mix is calculated from data on current crop composition in each region. Data on agricultural production patterns are collected from the Swedish Board of Agriculture and Statistics Sweden. The reason for using the typical crop mix is that biodiversity effects are defined as changes in the biodiversity on cereal fields that result from a change in the crop mix. For yield reductions, the cost is calculated as the reduced profit, taking into account adjustments made in inputs, and for conversion to organic farming, the cost is the conversion cost when management is changed, but products are sold as conventional. The maintenance and capital costs of machinery are treated as common costs and are not included in the estimated short-run production costs. For all cost calculations, agricultural support schemes are taken to be exogenous.

Costs are calculated from data in the agricultural management database Agriwise (www.agriwise.org), which is a tool for strategic economic planning within agricultural management, and based on agricultural and economic research from the Swedish Agricultural University of Sciences. This tool permits the user to calculate the economic and production consequence of different management choices. The contribution margin for different crops and regions is obtained from the Agriwise database, and is defined as the per-hectare crop revenue minus the specific costs for seed, fertilizers, feed, and fuel for agricultural machines. Because of the large variation in agricultural prices over the last years, costs were calculated for actual prices in 2011 as well as for the average prices 2008-2010. OECD and FAO both predict that the cereal prices will increase in the next decade by 10 to 30 per cent, due to population growth spurring increased demand, and increased use of biofuels (Board of Agriculture, 2010b). Swedish cereal prices have for the last years followed international price developments. Given that 2011 prices seem to better reflect expected future prices, wherefore we use those in the following[5].

Study area

The study is applied to GSS and SS, which are the two dominant agricultural plain regions in Sweden. The choice is motivated by the rapid decrease in biodiversity in these regions due to the structural changes in the agricultural production in combination with the comparatively low adoption of AEMs. The latter is largely explained by the high profitability of agricultural production in these regions in combination with the use of uniform compensation schemes for AEMs across the country.

The two regions differ substantially with regard to land use. In GSS, the predominant land cover is agricultural land, 42 % compared to 19 % in SS, while forest is the most prevalent land cover type in SS, 62 % compared to 36 % in GSS. In both regions, autumn wheat is the dominant cereal crop, covering 33 % of arable land in GSS and 19 % in SS, followed by spring barley. Ley cultivation is twice as abundant in SS compared to that of GSS, covering 32 % of the agricultural land compared to 14 % in GSS. Further, there are differences in climate and hence the length of the growing season as well as in management

[5] Average prices are included in the Appendix for comparison.

regimes. For example, pesticide and fertilizer use is higher in GSS (Board of Agriculture, 2008), which affects e.g. sward height and density, and therefore also has an impact on both flora and fauna (Wilson et al., 2005; Hiron et al., 2012). Data on land-use and yields are available in the Appendix. Overall, SS is characterized by a mosaic mixture of arable land, pastures, and forest, while GSS is dominated by a more productive, intensively farmed, and consequently a more homogenous agricultural landscape (see Figure 2).

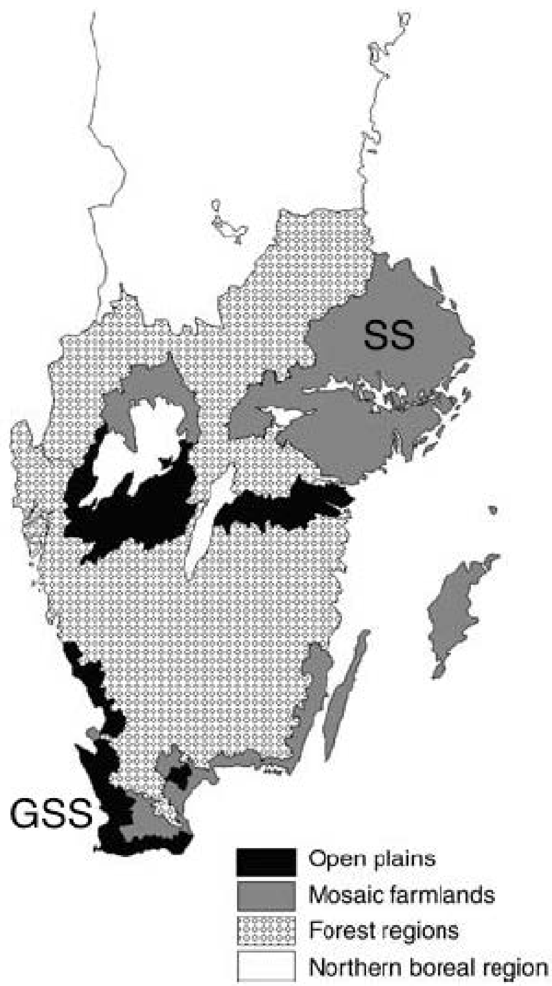

Figure 2. Swedish Landscape Structures (adapted from Wretenberg et al., 2007).

Costs and effects of different measures

In the following, the costs and biodiversity effects of the different measures are described. The choice of measures to include is based on the need for having, simultaneously, data on expected effect, standard deviation of the effect and unit costs. A review of the literature showed that the measures for which all these data are available, or can be calculated, are: changes in mean field size, reduced crop yield, increased pasture cover, increased use of field margins, conversion to organic farming, and increased landscape heterogeneity.

Increased landscape heterogeneity

The agricultural landscape heterogeneity is important to biodiversity. One commonly used measure of biological heterogeneity is the Shannon Diversity Index (SDI). When applied to the landscape mosaic, the SDI expresses the geometric mean of the proportional abundance of different biological habitats or land uses. The SDI is increasing in the number of habitat types and increases when habitats types occupy more equal proportions of the landscape. The smallest value of the index is obtained when the whole landscape consists of one habitat type and the largest value is obtained when the landscape is distributed equally among many different habitat types[6]. The index, here denoted *H´*, is calculated as:

$$H' = -\sum_{i=1}^{n} p_i \log p_i,$$

for cover types i, with $i=1,\ldots,n$, in the landscape of interest, and p_i is the share of habitat type i, i.e.:

$$p_i = \frac{area\ of\ habitat\ type_i}{\sum_{i=1}^{n} area\ of\ habitat\ type_i}$$

The choice of log base is standardized as the natural logarithm (Magurran, 2004). The calculation of SDI follows Geiger et al. (2010) by including eight different cover types: urban fabrics, cultivated arable lands, fallow lands under rotation systems, transitional woodland scrub, permanent crops, pastures, forests, and water. For their observations, a mean SDI of 0,6, and a standard deviation of 0,4, is obtained. Species richness is shown to be significant, and positively related to landscape heterogeneity, but no significant impact on bird abundance is found in Geiger et al. (2010).

In order to calculate the cost of increasing landscape heterogeneity, we first calculate the current SDI for GSS and SS, while including the same habitat types as above. Land uses in the regions are calculated from data for municipalities belonging to these region, taking into account the share of each municipality that belongs to the region in question (SCB, 2009). Data are available in the Appendix. Data on permanent crops and transitional woodland are not available as these land uses are very uncommon in the study regions, wherefore the area is assumed to be zero.

If all land uses could be altered, then with eight land uses the SDI is maximised if all habitat types cover the same fraction (1/8) of the total land area. The highest attainable SDI would then be 0,903, while

[6] Even though the Shannon index is widely used, it may be biased if not all habitat types are included in the index (Southwood and Hendersson, 2000; Magurran, 2004). Yet this is less important in this context, where the purpose is to investigate potential changes in the index, and comparing such changes between regions.

the lowest, when the landscape consists of only one habitat type, approaches 0: *H´*=-(1*ln(1)+0+0+…+0), where the first term goes to zero.

Under current land use, the SDI of GSS, 0,512, is higher than that of SS, which is 0,463, which could be somewhat counterintuitive given the strong role of agriculture in GSS. However, a great part of the difference is caused by the fact that the amount of forest in SS is about 7.5 times larger than in GSS. It should be noted that the potential to increase heterogeneity through changes in land use cropon arable land is larger in GSS, due to the larger homogeneity of arable land in that region, which is the factor of interest for this study. Here, we investigate only changes in the agricultural landscape, and thus only allow for changes in arable land, fallow land, and pasture. In that case, the largest possible SDI which could be achieved through a changed land use mix is 0,672 in GSS and 0,505 in SS. Figure 3 illustrates the theoretical and regionally achievable maximum value as well as current SDI values in GSS and SS.

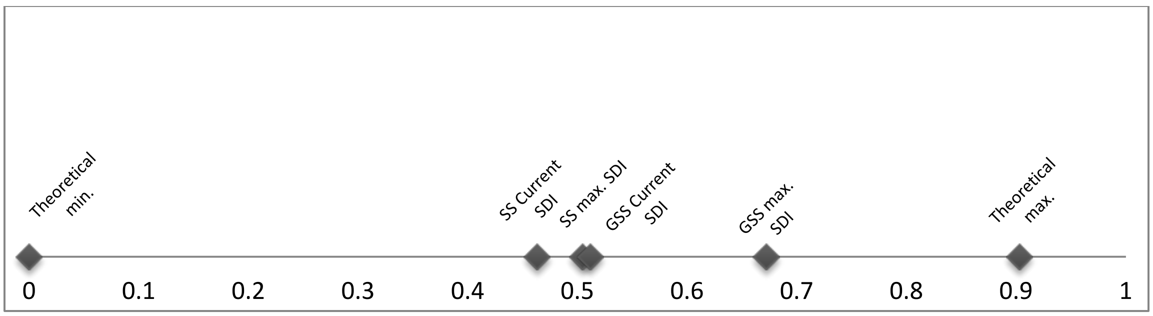

Figure 3. Theoretical maximum and minimum, current, and achievable values of SDI in GSS and SS regions.

Next, we calculate the area-weighted net revenue per hectare in a region as:

$$\pi_{region} = \sum_{i=1}^{H} \pi_i * p_i$$

in which π_{region} represents average per hectare net revenue, π_i is the per hectare profit on land use *i* and p_i is the share of the land use *i* in the landscape.

Several cover types are assumed to be fixed, such as areas of water, urban land, and forests. Costs can thus only arise from changes in the agricultural land use. It is also assumed that due to crop rotation restrictions, the fraction of fallow land cannot decrease.

The cost of increasing the SDI is defined as:

$$TC = \sum_{i=1}^{n} \pi_{pi}^{0} * p_i^0 - \sum_{i=1}^{n} \pi_{pi} * p_i \, , i = 1, \ldots, 8,$$

where the total cost, *TC*, is defined as the difference between the average net revenue under current land use, H'_0 with $H'_0 = \sum_{i=1}^{n} \pi_{pi}^{0} * p_i^0$, and the average net revenue under an alternative land use allocation, H', with $H' = \sum_{i=1}^{n} \pi_{pi} * p_i$. The minimum total cost of achieving a targeted increase $\Delta H'^{*}$ in the SDI, where $\Delta H'^{*} = H' - H'_0$, is then defined by:

$$\min TC = \sum_{i=1}^{n} \pi_{pi}^{0} * p_i^0 - \sum_{i=1}^{n} \pi_{pi} * p_i \,, i = 1, \dots, 8$$

$$s.t.$$

$$\sum_{i=1}^{n} p_i \log p_i \; - \sum_{i=1}^{n} p_i^0 \log p_i^0 \; = H' - H'_0 = \Delta H'^{*}, i = 1, \dots, 8$$

$$p_{fallow} \geq p_{fallow\,0}$$

$$p_{forest} = p_{forest\,0}$$

$$p_{water} = p_{water\,0}$$

$$p_{urban\ land} = p_{urban\ land\,0}$$

We c calculate the marginal cost for an increase in SDI by 0,01 index units for each of the two regions. So calculated, the cost-effective change in land use is an increase in fallow land relative to pasture and arable land in GSS, whereas in SS the cost-effective land use change is to increase pasture relatively other agricultural land uses in SS.

A variety of studies confirm a positive effect of increased landscape heterogeneity on species richness (for an exception, see Smith et al., 2010). The corresponding magnitude of the impact is here calculated from the results in Geiger et al. (2010), where it is shown that a 0,01 increase in the SDI is associated with an increase in bird richness by 0,00025 species per hectare. The corresponding coefficient of variation is 0,20.

Decreased field size

Mean field size can be seen as one of several possible proxies for landscape heterogeneity (Belfrage et al., 2005; Rundlöf et al., 2008). Using data in Agriwise, we calculate the cost of a decrease in mean field size from 200 ha to 70 hectares, which are the two different field sizes available in the database. The difference in net revenues between fields of different size is then mainly determined by scale economies with regard to management and input use, whereas yield differences per hectare are small. Based on results in Geiger et al. (2010), the corresponding biodiversity effect is calculated for both bird abundance and richness, given that the impact is significant for both measures.

Yield reduction

Yield is frequently used as a proxy for the effect of agricultural intensity on biodiversity (Donald et al., 2001; Geiger et al., 2010) and estimations in Geiger et al. (2010) show that bird abundance and richness are both negatively related to yield. Yield is defined as the yield weight on a crop-area-weighted hectare of arable land used for crop production for each region. For these calculations, yield data from 2008-2010 are used to avoid bias due to variations in annual yields. Thereafter, the average regional yield is multiplied by the associated average net revenue for the same cereal mix to obtain the corresponding average net revenue. The corresponding impact on biodiversity is obtained from Geiger et al. (2010).

Pasture
The percentage cover of pasture is an indicator of farming intensity and of the importance of animal husbandry in the region (Geiger et al., 2010). Semi-natural pasture often contains many natural habitats and niches that sustain higher levels of biodiversity. The impact of increased pasture land on bird abundance and richness is obtained from Geiger et al. (2010). The cost of the measure is calculated as the opportunity cost of increasing the share of pasture by 10,2 per cent, compared to maintaining the current crop-mix in the regions.

Grassy field margins
In many EU member states, farmers can receive support for construction of grassy field margins. In Sweden, it is required that the field margins constitute a strip of at least 6 meters width, sown with grass, a mix of grass and leguminous plants, or a mix of seeds that favour pollinating insect populations. The main motive behind the support is that grassy field margins retain nitrogen and herbicide leaching and so nearby lakes and streams can be protected from pollution, while another one is the possible impact on biodiversity, where the latter is little studied (Smith et al., 2011; Hof and Bright, 2010).

Here, the cost of grassy field margins is calculated as a conversion to fallow on the field margin in question on a square field of with the regionally typical crop composition. We assume that the field is completely surrounded by field margins, as when the purpose is biodiversity enhancement, it is not obvious that proximity to water is of importance. The proportion of the field covered by field margin differs depending on field size and shape and is calculated according to:

$$Field\ margin\ share = \frac{Field\ area - \ Crop\ Area}{Field\ Area}$$

where *field area* is width times length, and *crop area* is the total area minus the margin area. Effectively, the proportion covered by the field margin is smaller on a 200 ha field compared to a 70 ha field. In addition, the field shape can affect the proportion that constitutes field margin; a long narrow field has a greater proportion of field margin due to its longer circumference compared to a quadratic squared field with the same area (see figure 4).

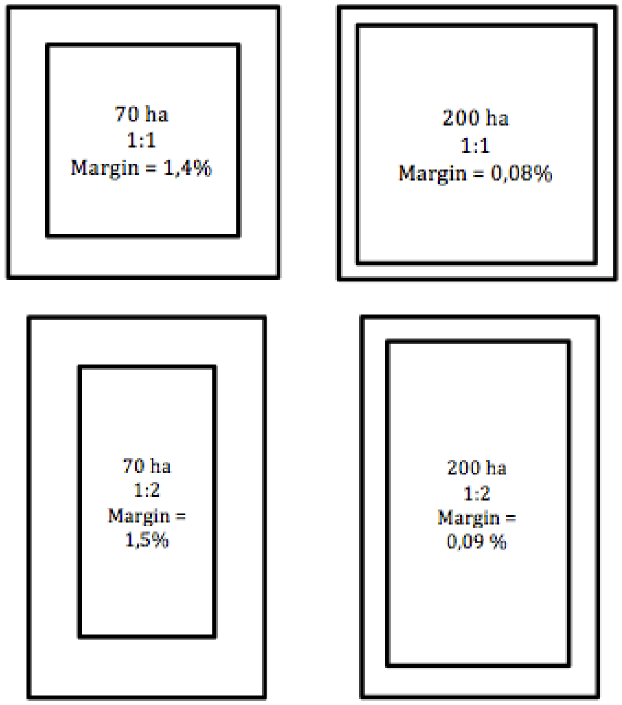

Figure 4. Field shapes and the share allocated to field margins when the field margin is 6 m wide.

There are no available statistics on field shapes. Since a field with the ratio between length and with equal to 1:4 is the most profitable shape from an agri-economic point of view (Hallefält and Nilsson, 2006), we subjectively assume that the most profitable 1:4 fields make up 60% of the total field shapes in the regions, and that the proportions 1:1 and 1:2 make up 20% respectively. The average field shape obtained is henceforth be referred to as the *typical field shape*. For the typical field shape, a six-meter wide field margin on a 70-hectare field then occupies 1,16 hectares, and on a 200-hectare field occupies 1,97 hectares. We calculate the cost based on yield losses on a typically shaped 200-hectare field, assuming that management is identical to that on managed fallow fields, i.e. grass is sown, and some management activity is necessary.

For biodiversity effects of grassy field margins, only effects on the abundance of skylarks is available as this data comes from a separate study (Josefsson et al., *in prep.*). However, since the skylark (*Alauda arvensis*) is one of the farmland bird species that has declined most, and is a so called flagship species of agricultural land, it is of interest to identify measures that could halt the decline. The effect on bird abundance of grassy field margins are estimated from data collected in SS. Inventories were carried out in 2012 and included fields with and without grassy field margins. In the plot selection process, fields with and without a grassy field margin were paired across multiple criteria, such as crop type (winter/spring sown varieties of oat, barley, wheat and rapeseed), field size, distance to forest and size of ditch bordering the field) to account for other factors potentially affecting skylark abundance. In total 22 fields were sampled, with 10 spring sown and 12 autumn sown fields.

Skylarks were counted five times in intervals of one week between May 22th and June 21st. Visits were made in good weather between 8 a.m. and 3 p.m. and the timing of visits was randomised so that no fields were sampled only in the mornings or only in the afternoon. Each observation lasted for five minutes and was made from the field border. Study plots extended into the field as a semi-circle with a radius of 100 meters. In this period, the location and behaviour (singing, foraging etc.) of skylarks were recorded as well as any additional bird individuals belonging to other species.

To derive an effect size for the effect of grassy field margins on skylark abundance, the data was analyzed with a Generalized Linear Mixed Model (GLMM) using the lme4 package in R. Explanatory variables were presence of grassy field margin (nominal variable, presence/absence) and time of visit (to account for seasonal differences in effect size) and were included as fixed effects. To account for sampling structure, we included Field nested in Region as random effects. The number of singing skylarks in was modelled with Gaussian errors and identity link function. The effect size of grassy field margins was determined to an increase of 0.20 skylarks per hectare, with standard deviation 0.11, in plots with grassy field margins compared to plots without grassy field margins[7].

Conversion to organic farming

Organic farming is usually understood as production methods that preserve natural resources and increase biodiversity (Winqvist et al., 2012; Diekötter et al., 2010; Geiger et al., 2010). Although there is some variation in the precise definition of organic farming across countries, organizations, and support schemes, organic farming generally implies zero use of pesticides and chemical fertilizers, and frequent crop rotation (Hole et al., 2005). In a review of 76 studies, Hole et al. (2005) conclude that organic farming is clearly related to higher species abundance and richness compared to conventional farms. Given the multiple production changes implied by organic farming compared to conventional, the importance of the individual production changes for the effect on biodiversity are not fully known (Rundlöf et al., 2008).

In order to qualify for organic farming certification, the soil must be pesticide-free for a certain time, which varies depending on previous management and crops cultivated. During such a qualifying period, the farmers receive the same product prices as conventional producers, and are eligible for agri-environmental support to organic farming. Once the farm is certified, it receives higher product prices and a higher support (Board of Agriculture, 2012). We calculate the short-run costs as the difference between net revenues under the conversion period with those at a conventional farm given a regional typical crop-mix yield. The yield reduction for each crop under the conversion period is assumed equal to the actual difference between conventional and organic farms yields in each region. The subsidy received during conversion is taken into account.

This approach implies that we do not take into account costs and revenue changes beyond the conversion period, such as e.g. certification costs, higher product prices and subsidy payments to certified organic farming. Although a simplification, it can be noted that future costs and revenues are of smaller importance to a farmer compared to those that occur in the near future if the farmer has a positive discount rate. It should be noted that the support to organic farming after conversion is partly motivated

[7] While this is the most tanglible data available for the effects of grassy field margins on bird abundance, there are several circumstances that need to be considered when interpreting the results of this study. First, while the grassy field margins attract skylarks, the effect on other bird species probably differ as a result of differing life histories and thus, it is difficult to draw any conclusions regarding response on overall abundance (i.e. aggregated response across all 29 farmland species) of the margin strips. Further, the inventories were only carried out close to the field margin itself where the effect of the strips is probably biggest. Thus, it is problematic to apply these results to landscape scale responses since the effect of the strip most probably only attract birds close to the strip itself.

by the impact of organic farming on biodiversity, which further motivates excluding it from the calculations as it is not exogenous to the problem studied.

Summary of costs and effects

Data on regional biodiversity baselines, percentage and absolute biodiversity effects, and regional costs is presented for richness in table 4 and for abundance in table 5. Where applicable, data for both conventional and organic farms is presented.

		Bird abundance baseline (no./ha)	Increase in bird abundance, % (st.dev.)	Absolute effect on bird abundance (no./ha)	St. dev. of abs. effect bird abundance	Cost (SEK/ha, 2008-2010 avg prices)	Cost (SEK/ha 2011 prices)
Yield decrease (Kg/ha)	GSS Conv.	0,80	0,08 (0,04)	0,06	0,03	452	1384
	GSS Org.	1,11		0,09	0,04	2378	2380
	SS Conv.	0,73		0,06	0,03	440	1270
	SS Org.	0,69		0,05	0,03	3872	2996
Farm type conversion (conventional to organic)	GSS Conv.	0,80	0,10 (0,05)	0,08	0,04	-	1908
	SS Conv.	0,73		0,07	0,04	-	950
Decreasing field size (decrease from 200 to 70 ha)	GSS Conv.	0,80	0,24 (0,02)	0,19	0,02	86	235
	GSS Org.	1,11		0,27	0,02	1078	115
	SS Conv.	0,73		0,18	0,02	95	226
	SS Org.	0,69		0,17	0,01	639	141
Pasture (conversion of 10,2% of arable land)	GSS Conv.	0,80	0,12 (0,05)	0,10	0,04	64	436
	GSS Org.	1,11		0,13	0,06	282	365
	SS Conv.	0,73		0,09	0,04	6	237
	SS Org.	0,69		0,08	0,04	247	315
Grassy field margin	GSS Conv.	0,80	NA	0,20	0,11	27	63
	GSS Org.	1,11	NA	0,20	0,11	58	58
	SS Conv.	0,73	NA	0,20	0,11	36	73
	SS Org.	0,69	NA	0,20	0,11	79	88

Table 4. Effects on abundance and costs for measures in different regions and for different farm types.

		Bird richness baseline (no./ha)	Increase in bird abundance, % (st.dev.)	Absolute effect on bird richness (no./ha)	St. dev. of abs. effect bird richness	Cost (SEK/ha, 2008-2010 avg prices)	Cost (SEK/ha 2011 prices)
Yield decrease (Kg/ha)	GSS Conv.	0,52	0,03	0,016	0,008	452	1384
	GSS Org.	0,96	0,0152	0,029	0,015	2378	2380
	SS Conv.	0,53		0,016	0,008	440	1270
	SS Org.	0,44		0,013	0,007	3872	2996
Farm type conversion (conventional to organic)	GSS Conv.	0,52	0,08	0,042	0,017	-	1908
	SS Conv.	0,53	0,0324	0,042	0,017	-	950
Decreasing field size (decrease from 200 to 70 ha)	GSS Conv.	0,52	0,0957	0,050	0,012	86	235
	GSS Org.	0,96	0,0233	0,092	0,022	1078	115
	SS Conv.	0,53		0,051	0,012	95	226
	SS Org.	0,44		0,042	0,010	639	141
Pasture (conversion of 10,2% of arable land)	GSS Conv.	0,52	0,05	0,026	0,011	64	436
	GSS Org.	0,96	0,0209	0,048	0,020	282	365
	SS Conv.	0,53		0,027	0,011	6	237
	SS Org.	0,44		0,022	0,009	247	315
SDI increase	GSS All	0,484	0,00025	0,00012	0,00005	31	32
	SS All	0,5041	0,0001	0,00013	0,00005	2	2

Table 5. Effects on richness and costs for measures in different regions and for different farm types. The SDI effect is evaluated at a 0,01 increase in the SDI index.

Results

In the results section, we first compare costs and expected effects measures in order to investigate whether these two different data sets give rise to different policy conclusions. This is relevant as current policies are, largely, designed based on limited, qualitative information about the impact on biodiversity in combination with limited information on costs, as typically, only national average costs are calculated. We therefore choose to investigate the implications of having limited information. Second, we compare this to the conclusions about the preferred choice of measures based on cost-effectiveness with or without inclusions of uncertainty about the biodiversity impact. Thereby, we can compare conclusions that would be drawn under limited information to those that would be drawn when both economic and ecological information is available. Third, we investigate whether specific patterns arise, which suggest that measures in one of the regions or on one of the farm types are associated with lower costs. Finally, sensitivity analysis is carried out in order to find out whether results are robust with respect to uncertainties in the data used.

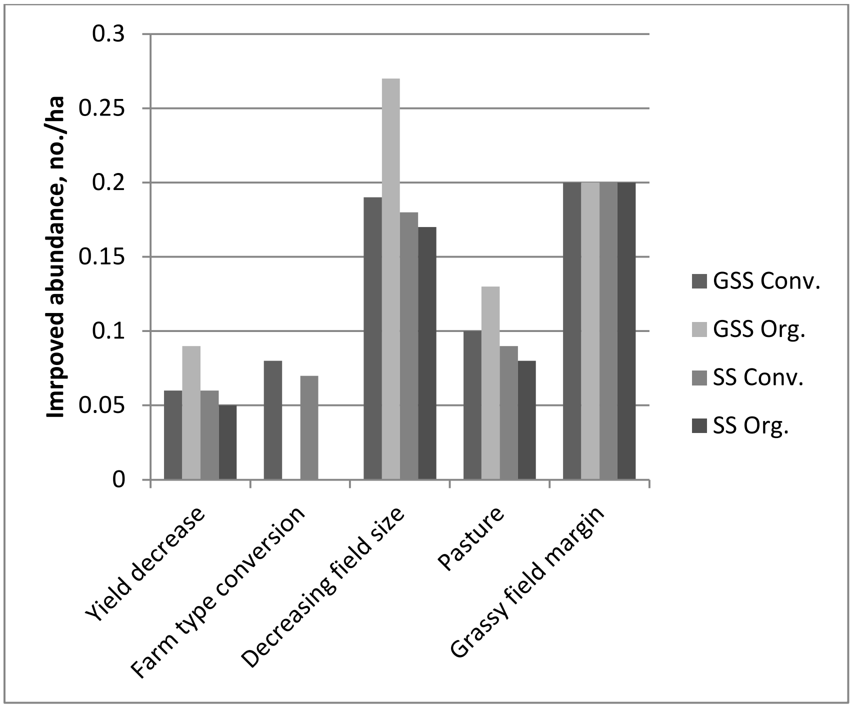

Figure 5a

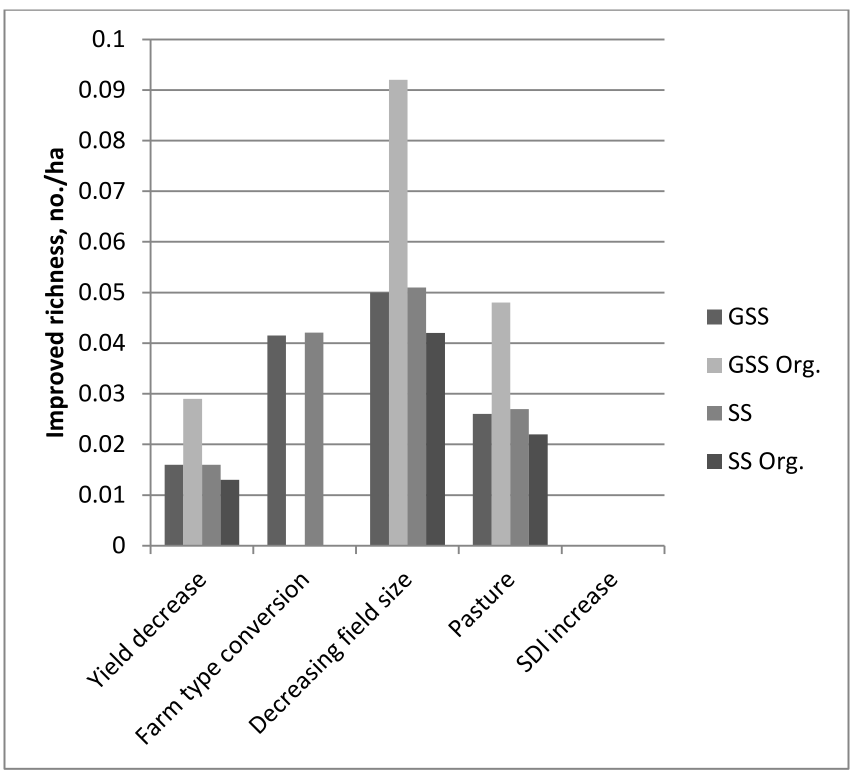

Figure 5b

Figure 5a and b. Effect of measures on bird abundance and bird richness per unit of measures implemented.

Does information on costs and effects have the same policy implications?

Figure 5a and b illustrate the unit impact of different measures on biodiversity, measured in terms of abundance and richness. As a rule of thumb, most measures that affect both biodiversity measures provides a three times larger expected impact on bird abundance compared to richness. Given the magnitude of change investigated for each of the measures, a reduction in field size on organic farms in GSS gives the largest effect on biodiversity in both cases. The second largest impact on abundance is obtained from grassy field margins followed by reductions in field size on conventional farms and organic farms in SS. For species richness, the second largest impact in from reductions in field size on conventional farms followed by increased pasture on organic farms in GSS.

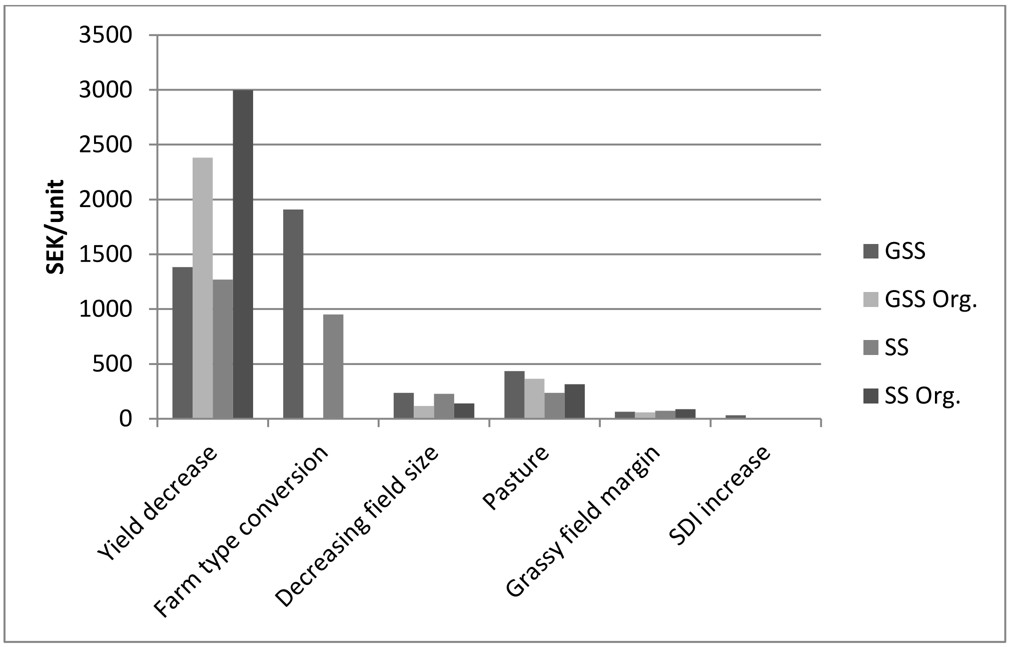

Figure 6. Unit cost of measures. (Units are defined in the text).

Figure 6 illustrates the corresponding unit cost for the different measures. An increase in SDI appears as the cheapest alternative, followed by, in turn, grassy field margins, reduction in field size and increased pasture area. Thus, similar policy conclusions would be drawn if decisions were made based on only one set of data, either ecological or economic, with an exception for increased SDI. An SDI increase by 0,01 units has a small impact and low cost, wherefore it would be disregarded based on only ecological information but could be chosen based on economic criteria. The reason is that it is difficult to judge how costs relate to the effect. We therefore turn to looking at cost-effectiveness.

Cost-effectiveness of measures

The cost-effectiveness of AEMs is judged based of the expected output in terms of improved biodiversity per unit of money spent. Figure 7a and b compare the cost-effectiveness of different measures in terms of the impact on the two bird abundance and richness per 1000 SEK. For both indicators, yield reductions and farm type conversion appear to deliver little biodiversity benefits per unit of money. When the target is to improve abundance, grassy field margins is, on the overall, the most cost-effective measure, followed by a reduction in field size. A reduction in field size is the most cost-effective measure to improve richness, followed by an increase in pasture area.

Cost-effectiveness of measures in different regions and on different farm types

There are differences in cost-effectiveness between measures in SS and GSS, see figure 7a and b. Given the method used and data available, differences in the biodiversity effect between regions is only determined by differences in the baseline biodiversity in the two regions. A consequence of the functional relationships estimated in Geiger et al. (2010) is that with a higher initial biodiversity, a higher effect is achieved. However, costs are also different between regions. Conversion to organic farming, increased pasture area and increased SDI is more costly in GSS – in all cases explained by the higher productivity of arable land in this region and hence the higher profits obtained under conventional farming. When ecological and economic factors are jointly taken into account, only conversion to organic farming and increased SDI are more expensive in GSS than in SS. It is worth noting that an increased SDI is the fourth most cost-effective measure in SS in order to improve richness, while it is the least cost-effective in GSS. The reason is that the opportunity cost of changing landscape heterogeneity by means of conversion of arable land to pasture in SS is very low, implying that even though the biodiversity effect is small, this measure gives a reasonably high value for the money.

Comparing measures undertaken at conventional and organic farms, a decrease in field size is more cost-effective at organic farms compared to conventional farms for both biodiversity indicators. This is mainly explained by smaller scale economies on organic farms compared to conventional, implying that the cost difference between different field sizes is smaller. For the other measures, the role of organic versus conventional farms varies between measures and regions. For example, increasing pasture area is more cost-effective on organic farms than on conventional farms in GSS with regard to both biodiversity indicators while the situation is the opposite in SS, the explanation being that the cost of converting arable land to pasture in GSS is higher on a conventional than on an organic farm, due to the higher productivity and revenue on conventional crop land. In SS, organic crop production is comparatively more profitable than in GSS, wherefore increasing pasture area is more cost-effective on conventional farms. Yield reductions are more expensive on organic farms in both regions, wherefore this measure is inefficient from a cost-effectiveness point of view compared to measures at conventional farms in both regions.

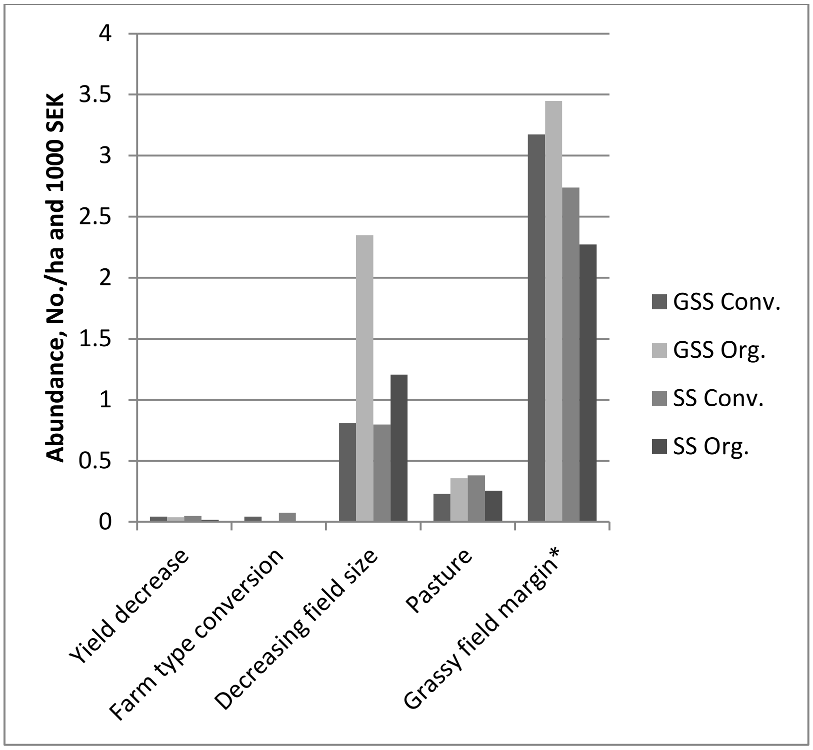

Figure 7a

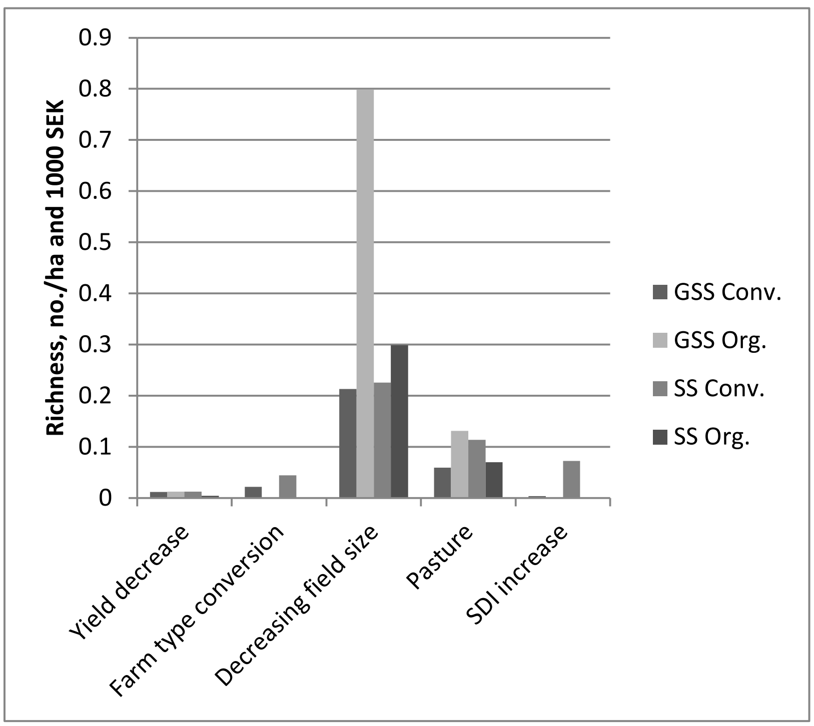

Figure 7b

Figure 7a and b. Cost-effective measures to improve bird abundance and richness.

The role of uncertainty for the preferred choice of measures

Next we investigate the role of uncertainty for the cost-effectiveness of the included measures. Table 6 shows the cost-effectiveness of different measures on bird abundance, measured as the certainty equivalent divided by the unit cost, under different certainty requirements. Note that a 50% certainty requirement is equivalent with placing zero value on uncertainty. Results show that with a higher certainty required, pasture and reductions in field size tend to become even more attractive measures than before, while grassy field margins become less attractive, although are still more cost-effective than yield decreases and farm type conversion.

	GSS Conv.	**GSS Org.**	**SS Conv.**	**SS Org.**
Yield decrease (50%)	0,043	0,038	0,047	0,017
Yield decrease (80%)	0,025	0,024	0,027	0,008
Yield decrease (95%)	0,008	0,010	0,009	0,000
Farm type conversion (50%)	0,042		0,074	
Farm type conversion (80%)	0,024		0,038	
Farm type conversion (95%)	0,008		0,005	
Decreasing field size (50%)	0,809	2,348	0,796	1,206
Decreasing field size (80%)	0,737	2,199	0,721	1,149
Decreasing field size (95%)	0,669	2,060	0,650	1,092
Pasture (50%)	0,229	0,356	0,380	0,844
Pasture (80%)	0,152	0,218	0,238	0,147
Pasture (95%)	0,079	0,087	0,103	0,046
Grassy field margin* (50%)	3,175	3,448	2,740	2,273
Grassy field margin* (80%)	1,710	1,858	1,478	1,224
Grassy field margin* (95%)	0,312	0,339	0,269	0,223

Table 6. Improvement in bird abundance, certainty equivalent (no/ha) per 1000 SEK under different certainty requirements.

In a corresponding manner, table 7 gives the cost-effectiveness of different measures on bird richness under different certainty requirements. Results show that if it is required that richness effects are achieved with 95% certainty, the certainty equivalent achieved per SEK spent falls by between 40 and 84 per cent, varying between the measures. With a higher reliability level, a reduction in field size becomes an even more attractive measure compared to the other measures than for lower levels of reliability. Note that for both abundance and richness, yield reductions become even less attractive when higher certainty is required, as this measure is not only costly but also associated with considerable uncertainty about the impact.

When taking certainty into account, the relative ranking of the measures according to cost-effectiveness is to a large extent preserved. In particular, one of the most cost-effective measures, reduced field size, becomes even more cost-effective relative the other measures due to the comparatively small standard deviation. When uncertainty is taken into account, reduced field size is everywhere preferred to grassy field margins. The little cost-effective yield reductions become even less cost-effective compared to the other measures due to the relatively high standard deviation. When the aim is to improve abundance, farm type conversion becomes equally cost-effective as yield reductions if 95% is required, whereas this is not the case for lower levels of certainty.

	GSS	**GSS Org.**	**SS**	**SS Org.**
Yield decrease (50%)	0,012	0,012	0,013	0,004
Yield decrease (80%)	0,007	0,007	0,008	0,002
Yield decrease (95%)	0,003	0,002	0,003	0,001
Farm type conversion (50%)	0,022		0,044	
Farm type conversion (80%)	0,014		0,034	
Farm type conversion (95%)	0,007		0,025	
Decreasing field size (50%)	0,213	0,799	0,225	0,299
Decreasing field size (80%)	0,172	0,636	0,182	0,237
Decreasing field size (95%)	0,134	0,481	0,141	0,178
Pasture (50%)	0,060	0,131	0,114	0,070
Pasture (80%)	0,040	0,085	0,076	0,045
Pasture (95%)	0,022	0,041	0,041	0,022
SDI increase (50%)	0,004		0,072	
SDI increase (80%)	0,002		0,049	
SDI increase (95%)	0,001		0,027	

Table 7. Improvement in bird richness, certainty equivalent no/ha and 1000 SEK under different certainty requirements.

Sensitivity analysis

In this section we make sensitivity analysis with regard to the expected impact of measures on biodiversity, in order to find out whether results are robust with regard to uncertainty about parameters used and assumptions made for the calculations. One potential source of error is assumptions made about differences in biodiversity impact across regions and farm types. Calculations made in this paper draw on the results in Geiger et al. (2010), implying that differences in impact are solely determined by differences in baseline biodiversity. Yet, it is possible that in effects vary across regions and farm types for other reasons.

To investigate the implications of assumptions made about the effect on biodiversity, we calculate results assuming that the expected impact of a measure on a given biodiversity indicator is equal across regions and farm types, and equals the unweighted average of the impacts in absolute terms as reported in tables 5 and 6[8]. The results of this analysis are shown in Figure 8a and b below. The most important change compared to the above results is a reduction in the cost-effectiveness of measures applied at organic farms in GSS, and an increase in the cost-effectiveness of measures applied at organic farms in SS. For abundance, pasture on organic farms in SS is now more cost-effective compared to pasture on conventional farms. Looking at richness, the equal-impact assumption implies that yield reductions at organic farms in SS outperform that at organic farms in GSS. Also, reductions in field size at organic farms in SS outperform those at conventional farms in both regions, and for pasture, the measure is now more cost-effective at both farm types in SS compared to organic farms in GSS.

[8] This is not to say, that it is likely that there is a uniform effect across regions and farm types, the purpose is merely to see whether results are robust with regard to the assumption that the differentiation in the effect, implied by results in Geiger et al. (2010), matters for the conclusions, compared the case with no differentiation. There is no alternative data on the differences in impact across regions and farm types available, which could be used to test robustness of the results.

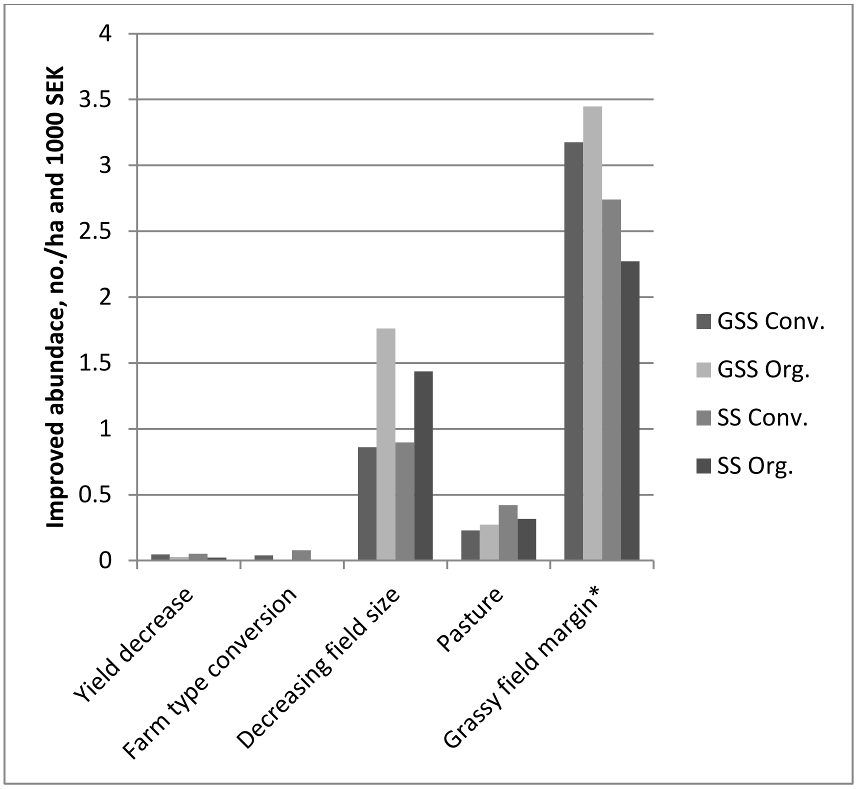

Figure 8a

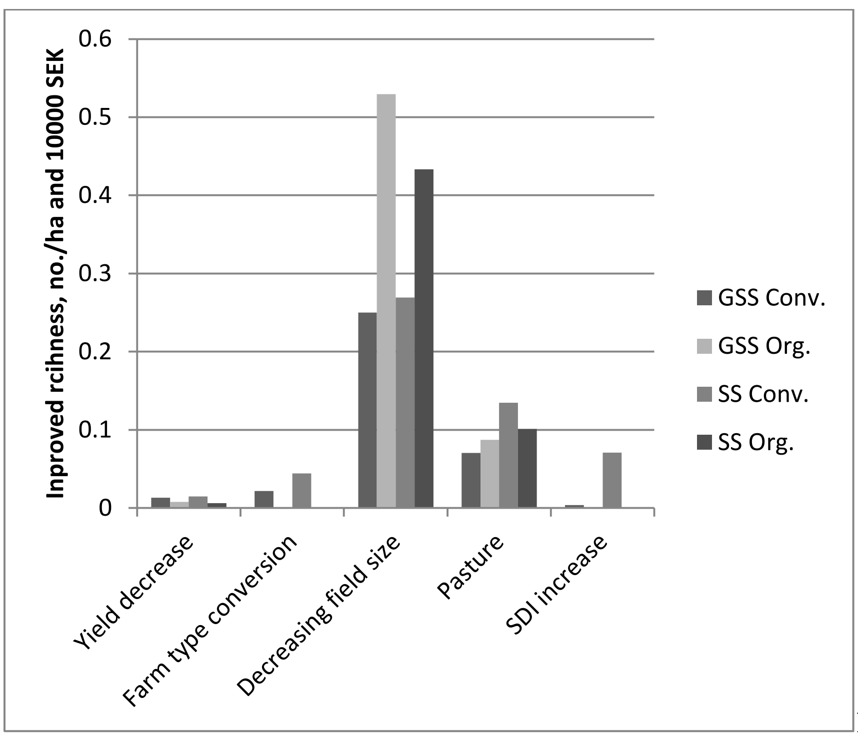

Figure 8b

Figure 8 and b. Cost-effectiveness of measures when impact on biodiversity is assumed identical across regions and farm types.

Discussion

The aim of this study is to compare the cost-effectiveness of different measures to improve biodiversity in the agricultural landscape, while taking into account uncertainty about the impact of measures on biodiversity. Uncertainty is accounted for through the calculation of a certainty equivalent of the impact of each measure, which is determined by the policy makers risk aversion, and the variance and distribution of the impact. The study is applied to improvements in bird richness and abundance through changes in agricultural practices in the Plain Districts in Svealand (SS) and Southern Götaland (GSS) in Sweden. Differences in cost-effectiveness between agricultural regions as well as between conventional and organic farming systems are examined.

Results show that reducing field sizes would be a cheap measure to improve bird abundance and richness, the same holds for increasing pasture area. Notably, reductions in field size are not subject to support through agri-environmental policy schemes, whereas natural grazing lands are. Introduction of grassy field margins would be the cheapest measure of all with regard to improvements in bird abundance. This measure is subject to support, although with the major aim to reduce nutrient leaching, which motivates the current requirement that grassy field margins are to be located in the proximity of water courses. Reductions in yield, conversion to organic farming and increasing the landscape heterogeneity are all comparatively expensive measures. Of those, organic farming is subject to support. Results here then indicate that for such support to be economically motivated, there must be either high ambitions to provide large improvements in biodiversity, where the large costs for the measure can be defended, or the support needs to be motivated by other benefits provided from organic farming. Notably, if the support is motivated by high ambitions to improve biodiversity, then also e.g. reduced field sizes should be subsidized if policy makers are concerned about costs. Results with regard to the cost-effectiveness of measures are robust with regard to uncertainty about their impact on biodiversity.

Some general conclusions can be drawn regarding the difference between regions and farm types. The most important is that conversion to organic farming, and increased landscape heterogeneity is more expensive in GSS compared to SS, due to the higher productivity of arable land and hence higher profits in this region. This suggests that if policy-makers have strong ambitions to increase biodiversity, such that also more expensive measures need to be included in the policy, these more expensive measures should be implemented in SS, provided that an increase in biodiversity in SS is considered equally valuable as in GSS. Reduced field size appears as a low-cost measure at organic farms due to the fact that there are smaller scale economies in cereal farming compared to conventional farms. Notably, sensitivity analysis suggests that the relative cost-effectiveness of, on the one hand, measures at organic farms in GSS, and measures at different farm types in SS on the other hand, are sensitive to assumptions about current bird richness and abundance.

As a reduction in field size turns out to be a cheap measure to increase bird abundance and richness, some discussion about the robustness of this conclusion is called for. At first glance, field size might be perceived as determined by landscape conditions and hence a factor than cannot be altered easily. However, there are several mechanisms behind the impact of field size on biodiversity, where the actual impact is confirmed by several ecological studies and that could be translated into feasible changes in management. First, smaller fields are likely to be associated with larger landscape heterogeneity due to e.g. larger crop variation, and the presence of border elements between fields, such as uncultivated margins. Therefore, the beneficial impact of reduced field size might potentially be achieved through division of larger fields into smaller while simultaneously introducing border elements or field margins that split up larger fields. The introduction of such border elements could increase the costs associated with the measure, due to yields losses and increased costs for management, wherefore the measure could potentially be less attractive than suggested by the results calculated above. However, even with a doubling of costs per hectare, results would not be radically changed, but reduced field size would perform equally well in cost-effectiveness terms as increasing pasture area. It should also be noted that a consequent introduction of reduced field sizes in combination with border elements could potentially lead to a larger impact on bird biodiversity compared to what is observed in ecological studies, given that a smaller field size in these studies is, most likely, sometimes associated with the presence of border elements, sometimes not. This would counteract the potentially higher cost caused by the introduction of border elements. Consequently, the results regarding this measure are reasonably robust with regard to the interpretation of the role of field size for costs and effects on biodiversity.

Pasture also appears to be a measure which could, at comparatively low cost, increase bird biodiversity. Notably, a larger area of pasture is shown in ecological studies to have a direct impact on bird biodiversity. We show that increasing pasture area would also be the most cost-effective way to improve landscape heterogeneity in SS. The impact on biodiversity through increased landscape heterogeneity is an additional benefit of the measure, which is not directly taken into account into the above calculations for the pasture measure, which would further strengthen the conclusion that pasture is a comparatively cheap measure in SS. It is also worth noting that pasture is more cost-effective on organic farms in GSS compared to conventional farms, whereas the opposite holds for SS. This could be explained by the more important role of organic farms for biodiversity in GSS given the otherwise homogeneous agricultural landscape, compared to SS, where the agricultural landscape is more heterogeneous.

Some of the measures indicated to be cost-effective are not included in current agri-environmental policy, and *vice versa*. This suggests that there is a need to evaluate the usefulness of the measures currently subject to support with a purpose to improve biodiversity in terms of their cost-effectiveness. Ultimately, this requires improved formulation of objectives, and further research that makes use of quantitative models to identify causal relationships between agricultural practices and biodiversity outcomes.

In the broader perspective, there are costs and benefits associated with the implementation of the measures discussed above, which are not accounted for in this study. Among the benefits are e.g. possible improvements in the biodiversity of other taxa, and possible improvements with regard to nutrient losses from agriculture. On the cost side, there are costs associated with implementation, which arise to farmers and implementing agencies, and that are not taken into account. This should be borne in mind when interpreting the results. In addition, the calculations made here only provide information on the costs and effects of a unit change in the measures investigated. The capacity of the different measures, and hence the potential role with regard to the overall ambitions to improve bird biodiversity has not been estimated within the project, but would require further research to be made.

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Appendix

Calculation of biodiversity effects

The coefficients obtained in Geiger et al. (2010) are standardised in order to make the effects comparable. In an OLS model, coefficients represent the rate of change of the dependent variable when the independent variable is changed by one unit, all other variables held constant. In order to fulfil the assumptions of a GLM (constant variance), each observation is standardised as:

$$\frac{X-\mu}{\sigma}$$

This means that the coefficient is representing the effect of a unit change of the standardised variable. This unit-standardised change is translated into an actual observable change by the standard deviation:

$$\frac{X-\mu}{\sigma} = 1 = \frac{\sigma}{\sigma}$$

$$X-\mu = \sigma$$

$$\hat{\sigma} = \sigma$$

$$X = \hat{\sigma} + \mu$$

So, in order to obtain the estimated coefficient change on abundance or richness, X needs to change with the estimated standard deviation of that variable. The actual effects are calculated according to the baseline.

The standard deviations of the biodiversity effects are calculated from the individual F-values. Assuming that Hotelling's T-squared distribution holds ($T^2_{p,n-1} \sim t^2$), and that the constant is close to 1, $t^2 \approx F$ and we obtain the standard deviation by:

$$\sqrt{F} = t = \frac{\beta}{\sigma}$$

$$\sigma = \frac{\beta}{\sqrt{F}}$$

GSS	Total Acreage 2010	% of Total Acreage	Organic Acreage	% of Total Acreage	Avg Region Yield 2008-2010	Prop. of Avg Region Yield	Avg Region Org. Yield 2008-2010	Prop. of Avg Region Org. Yield
Autumn Wheat	85212	33%	480	0,19%	584833	0,39		
Spring Wheat	9558	4%	410	0,16%	40800	0,03	1667	0,27
Rye	5555	2%	310	0,12%	45867	0,03	1200	0,12
Autumn Barley	4216	2%	20	0,01%	21800	0,01		0,00
Spring Barley	58986	23%	740	0,29%	390133	0,26	2533	0,41
Oat	8706	3%	270	0,11%	54067	0,04	1267	0,20
Triticale	5210	2%	310	0,12%	31233	0,02		
Mixed seed	353	0%	120	0,05%		0,00		
Peas and Broad Bean	2136	1%	300	0,12%	6800	0,00		
Rapeseed	33761	13%	150	0,06%	5800	0,00		
Linseed	712	0%	0	0,00%	109400	0,07		
Potatoe	5622	2%	100	0,04%		0,00		
Ley	34990	14%	2670	1,05%	207433	0,14		
SUM	**255017**		**5880**					

Table A1. Crop land distribution and average crop yield in GSS.

GSS		Revenue per hectare 2011 Prices				Revenue per hectare 2008-2010 Prices			Average Yield per hectare, 2008-2010	
Crop	**Conv 70 ha**	**Org 70 ha**	**Conv 200 ha**	**Org 200 ha**	**Conv 70 ha**	**Org 70 ha**	**Conv 200 ha**	**Org 200 ha**	**Conv**	**Org**
Autumn Wheat	6287	7428	6540	7693	907	4970	1010	5679	7397	0
Spring Wheat	4601		4811		3774		3840		5437	3100
Rye	5286		5545		3010		3113		6993	3895
Autumn Barley	4389		4631		2587		2319		6060	0
Spring Barley	3984	8556	4197	8745	2367	6620	2450	6672	5490	3610
Oat	2980	6269	3189	6456	1653	5428	1736	6503	4897	3187
Triticale	3913		4118		2206		2268		6067	
Mixed seed									0	
Corn									5930	

Peas	2236		2441		1440		1517		2910	
Broadbeans	2236				1440				2690	
Autumn Rapeseed	9058		9255		5544		5642		3827	
Spring Rapeseed	3272		3387		1841		1819		1670	
Linseed									0	
Potato	40745		40745		43590		43590		32333	
Pasture	1130	1130	1130	1130	1137	1137	1137	1137		
Fallow	-775				-759		-759			
Typical Crop Yield or Revenue SEK/ha	10399	4779	10596	4894	7802	3811	7875	4045	9901	3422
Typical Cereal Mix Yield or Revenue SEK/ha	5169	4779	5403	4894	1677	3811	1763	4045	6500	3422

Table A2. Average revenue per hectare and average yield per hectare in GSS.

SS	**Total Acreage 2010**	**% of Total Acreage**	**Organic Acreage**	**% of Total Acreage**	**Avg Region Yield 2008-2010**	**Prop. of Avg Region Yield**	**Avg Region Org. Yield 2008-2010**	**Prop. of Avg Region Org. Yield**
Autumn Wheat	85221	19%	4980	1,10%	483867	0,35	18500	0,27
Spring Wheat	32769	7%	4270	0,94%	101433	0,08	9800	0,15
Rye	3579	1%	370	0,08%	24800	0,02	1667	0,02
Autumn Barley	623	0%	0	0,00%		0,00		
Spring Barley	84206	19%	3940	0,87%	406433	0,29	7700	0,11
Oat	49915	11%	8780	1,93%	213100	0,15	22033	0,32
Triticale	4966	1%	620	0,14%	34400	0,02	2233	0,03
Mixed seed	3751	1%	1220	0,27%	13833	0,01	4967	0,07
Peas and Broad Bean	12608	3%	2280	0,50%		0,00		
Rapeseed	27811	6%	660	0,15%	18833	0,01	3100	0,02
Linseed	250	0%	10	0,00%	43100	0,03		
Potatoe	1352	0%	50	0,01%	4967	0,00		
Ley	147000	32%	40480	8,92%	33633	0,02		
SUM	**454051**		**67660**					

Table A2. Crop land distribution and average crop yield in SS.

SS	Revenue per hectare 2011 Prices				Revenue per hectare 2008-2010 Prices				Average Yield per hectare, 2008-2010	
Crop	**Conv 70 ha**	**Org 70 ha**	**Conv 200 ha**	**Org 200 ha**	**Conv 70 ha**	**Org 70 ha**	**Conv 200 ha**	**Org 200 ha**	**Conv**	**Org**
Autumn Wheat	4151	7313	4409	7523	156	4947	274	5602	5317	3177
Spring Wheat	3512		3718		3216		3282		4427	2617
Rye	3703		3958		1908		2015		4863	2607
Autumn Barley									0	0
Spring Barley	2780	6585	2991	6776	1548	5096	1635	5147	4097	2360
Oat	1967	4507	2176	4694	1224	3942	1308	4752	3853	2240
Triticale	3416		3619		1748		1812		4700	3100
Mixed seed									4160	2570
Corn										
Peas	1362	4779	1555	4634	864	4547	936	4281	2340	1637
Broadbeans	1425	3464	1610	3522	1558	4541	1643	4466	2150	1990
Autumn Rapeseed	4469		4773		2590		2775		2257	
Spring Rapeseed	4010		4120		2447		2422		1817	
Linseed	2582		2773		1799		1876		1453	
Potato	33776		33776		36172		36172		24060	
Pasture	1130	1130	1130	1130	1137	1137	1137	1137		
Fallow					-759		-759			
Typical Crop Yield or Revenue SEK/ha	3983	4241	4200	4378	2013	3260	2103	3698	4930	2614
Typical Cereal Mix Yield or Revenue SEK/ha	3230	4238	3456	4379	1102	3233	1198	3682	4578	2629

Table A3. Average revenue per hectare and average yield per hectare in SS.

Sånglärka, Skylark (*Alauda arvensis*)
Buskskvätta, Whinchat (Saxicola rubetra)
Stare, Starling (Sturnus vulgaris)
Gulsparv, Yellowhammer (Emberiza citrinella)
Ladusvala, Barn Swallow (Hirundo rustica)
Tofsvipa, Northern Lapwing (Vanellus vanellus)
Sädesärla, White Wagtail (Motacilla alba)
Pilfink, Tree Sparrow (Passer montanus)
Stenskvätta, Northern Wheatear (Oenanthe oenanthe)
Törnsångare, Whitethroat (Sylvia communis)
Hämpling, Linnet (Carduelis cannabina)
Fasan, Pheasant (Phasianus colchicus)
Steglits, European Goldfinch (Carduelis carduelis)
Gråsparv, House Sparrow (Passer domesticus)
Hussvala, House Martin (Delichon urbicum)
Näktergal, Nightingale (Luscinia megarhynchos)
Ängspiplärka, Meadow Pipit (Anthus pratensis)
Storspov, Eurasian Curlew (Numenius arquata)
Gulärla (sydlig), Western Yellow Wagtail (Motacilla flava)
Törnskata, Red-backed Shrike (Lanius collurio)
Enkelbeckasin, Snipe (Gallinago gallinago)
Tornfalk, Kestrel (Falco tinnunculus)
Vaktel, Quail (Coturnix coturnix)
Gräshoppsångare, Grasshopper Warbler (Locustella naevia)
Rapphöna, Grey Partridge (Perdix perdix)
Ortolansparv, Ortolan Bunting (Emberiza hortulana)
Kornknarr, Corn Crake (Crex crex)
Ängshök, Montagu’s Harrier (Circus pygargus)
Kornsparv, Corn Bunting (Miliaria calandra)

Table A4. Bird species used in calculation of baseline biodiversity.