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Changes in local species richness of farmland birds in relation to land-use changes and landscape structure

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ABSTRACT

It has been suggested that an increase in the area of low-intensity land-use on arable land (e.g. set-aside fields and short-rotation coppice), and high or increased farmland habitat heterogeneity, may halt or reverse the observed population decline of farmland birds. We tested these hypotheses by undertaking farmland bird censuses during two contrasting periods of agricultural policies and land-use (i.e. 1994 and 2004) in a farmland region covering a gradient of forest- to farmland-dominated landscapes in Sweden. Local species richness (i.e. at 3 hectare sites) declined significantly between 1994 and 2004. Local species richness was positively related to habitat heterogeneity in both years of study whereas temporal change in species richness was not. Local change in species richness was positively associated with a change in the proportion of non-rotational set aside and short-rotation coppice (i.e. low-intensity land-use forms), but also to changes in the amount of spring-sown crops. However, the effect of low-intensity land-use was significantly dependent on the amount of forest in the surrounding landscape. An increase in low-intensity land-use was linked to an increase (or less marked decrease) in species richness at sites located in open farmland surroundings but to a decrease in richness at sites located in forest surroundings. This interaction between amount of forest and low-intensity land-use could be interpreted as a "rare habitat effect", where an increase in a farmland habitat only positively affects biodiversity when it was originally uncommon (i.e. open farmland areas). Our results suggest that conservation measures of farmland biodiversity have to be put in a landscape context.

1. Introduction

The widespread and severe decline of farmland bird populations throughout Europe (Tucker and Heath, 1994; Siriwardena et al., 1998; BirdLife International, 2004; Wretenberg, et al., 2006) has generally been attributed to the intensification of agriculture (Chamberlain et al., 2000; Donald et al., 2001; Gregory et al., 2005). This agricultural change includes, for example, increased use of fertilizers and pesticides, simplified crop rotations, more homogeneous and dense crops and loss of semi-natural grasslands and non-farmed habitats (Stoate et al., 2001; Robinson and Sutherland, 2002; Newton, 2004; Wilson et al., 2005). As a result of these changes, habitats, food supplies and nesting sites have been lost or degraded in quality for many farmland bird species (Newton, 2004). For example, corncrakes (for scientific names, see Appendix 1) have declined because of reduced reproductive success due to changes in grassland management (e.g. earlier mowing dates; Green et al., 1997; Tyler et al., 1998). Skylarks may have suffered reduced reproductive success because of increased use of more dense and homogenous crops which are unsuitable as breeding and foraging habitat (Wilson et al., 1997; Chamberlain and Crick, 1999). Cirl buntings *Emberiza cirlus*, which feed on stubble fields during winter and on semi-natural pastures during summer, may have declined because of a simultaneous loss of mixed farming and a decline of winter stubble fields (Evans, 1997).

Countermeasures to halt the widespread decline are in many cases species-specific (Aebischer et al., 2000). However, two major measures have been suggested to cause a general positive effect on farmland bird communities, namely increased habitat heterogeneity (Benton et al., 2003) and reduced farming intensity (Krebs et al., 1999; Donald et al., 2006). First, an increased habitat heterogeneity, from the spatial scale of fields (e.g. by increasing the number of crops grown per farm) to whole landscapes is likely to have a broad positive effect on a number of species, since more species will be able to find suitable alternative habitats (Benton et al., 2003). Second, increasing the amount of set-aside fields or other low-intensity crops (e.g. short-rotation coppice) will increase availability of food and protected nest sites (Evans, 1997; Buckingham et al., 1999; Henderson et al., 2000a; Berg, 2002a) because

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these habitat types are cultivated less often and treated with lower levels of pesticides and fertilizers than other high-intensity crops (e.g. cereals, rape and pea).

We made two repeated large-scale censuses (1994 and 2004) of farmland birds in a Swedish farmland to evaluate the effects of habitat heterogeneity and low-intensity farming on farmland bird species richness. The agricultural policy shifted dramatically between these years; 1994 was the last year of an eight year period promoting low-intensity farming because of overproduction (i.e. the set-aside period) whereas farming practices and land-use in 2004 reflected a nine year period of increased production according to the Common Agricultural Policy in the European Union. Thus, we expected land-use to change markedly between the two censuses. First, we investigated whether local species richness of farmland birds was related to amount of low-intensity land-use (i.e. set-aside fields, short-rotation coppice, leys and cultivated pastures) and level of habitat heterogeneity. Second, we tested whether temporal changes in local species richness of farmland birds were related to corresponding changes in the amount of low-intensity land-use and whether such effects were linked to habitat heterogeneity in the surrounding landscape. For example, one may expect high habitat heterogeneity in the surrounding landscape to have a buffering effect on negative effects of changed local land-use on change in species richness because of a high availability of alternative habitats in the close surroundings. Third, we tested whether changes in species richness was linked to amount of forest in the surroundings because a previous study at the national scale suggested population declines of farmland birds to be similar in the large farmland plains and the forest-dominated farmlands of Sweden (Wretenberg et al., 2007).

2. Methods

2.1. Selection of census points

We used 212 census points (i.e. sites) located in the counties of Uppland and Västmanland (approximately 59°40′N–60°07′N and 16°30′E–18°10′E) in south-central Sweden (total area c. 1800 km^2). The sites were located in landscapes with differing amounts of forest (e.g. median 28%, range 0–92% as measured within a 600 m radius from the census point). However, all sites were located in farmland and the proportion of farmland within 100 m (i.e. the radius used for bird censuses) was high; 82% of the sites had more than 80% farmland within 100 m. The sites were mainly located in arable fields and the proportion of semi-natural pastures was low (4% within 300 m radius). All census points were located at least 600 m apart (median 900 m, range 600–4000 m). In 1994, the census points were selected in a stratified design with respect to different land-use types (i.e. spring-sown crops, autumn-sown crops, leys, cultivated pasture, set-aside fields and short-rotation coppice), semi-natural pastures, occurrence of residual habitats (e.g. ditches, within-field habitat islands and field roads) and landscape structure (forest-dominated and farmland-dominated). Initially, several hundred potential census points were investigated to cover variation in land-use. Almost all sites with short-rotation coppice were chosen (i.e. the rarest land-use) whereas the selection of the other census points was random within these strata.

Habitats were mapped within 300 m of the point centres with the help of field visits, land-use maps (1:10,000) and aerial photographs. Proportions of different habitats and land-use types (see Table 1) were estimated from these detailed maps using the software ArcView, version 3.3 (Anonymous, 1992–2002) with the XTool extension (DeLaune, 2001). Furthermore, we calculated three different heterogeneity indices: (i) length of linear habitats (i.e. total length (m) of all roads, ditches and grass strips between fields), (ii) landscape heterogeneity (i.e. number of transitions between arable land and other habitats) and (iii) land-use heterogeneity (see Table 1 for description of which variables were included in each index). Land-use heterogeneity was calculated according to Hurlbert's (1971) diversity index (i.e. "*PIE*", see Eq. (1)), where N equals the total number of land-use types and $p(i)$ represents the proportion of the entire sample represented by land-use type (i).

$$PIE = \left(\frac{N}{N-1}\right)\left(1 - \sum_{i=1}^{S} p_i^2\right) \tag{1}$$

We used the proportion of the landscape covered by forests within 300 m radius from the census point as our descriptor of the amount of forest in the landscape. This descriptor was strongly correlated with the proportion of arable fields at the same spatial scale ($r = -0.89$, $P < 0.001$) and with the proportion of forest at larger spatial scale (e.g. radius of 600 m; $r = 0.92$) and is referred to as "amount of forest".

2.2. Bird censuses

Birds were censused with point counts (Bibby et al., 1992). All sites were visited six times in 1994 and five times in 2004 during early morning (mainly from sunrise to 10 am), once in each of the periods 15–30 April (only 1994), 1–10 May, 11–20 May, 21–31 May, 1–10 June and 11–20 June. The first period (15–30 April) was excluded in 2004. However, this did not affect our local estimates of species richness since all species seen during this period were also seen at later visits in 1994. The sites were combined to groups of 11–23 sites which were visited during the same morning. Sites were censused in a different order at each occasion to avoid bias due to variation in diurnal activity of birds.

Birds were censused within a radius of 100 m of the census point. However, habitat structure and composition were mapped within a radius of 300 m (see above) because several species have large home ranges or are known to respond also to surrounding habitats (see e.g. Söderström and Pärt, 2000). The observer noted all bird species seen and heard during five minutes. Birds that were flushed when approaching the point and observed within the distance limit were also included. No counts were made on mornings with strong wind or rain. Observations of singing males and pairs were used to estimate the number of species at each site. We included 41 bird species in the estimate of species richness (see Appendix 1), which we considered to be farmland bird species according to previous studies from this region (Berg and Pärt, 1994; Pärt and Söderström, 1999a,b; Söderström and Pärt, 2000; Berg, 2002b). However, flocks of obviously migrating birds were not included in the censuses.

2.3. Statistical methods

Associations between species richness (i.e. the total number of species with territories at each site) and habitat variables were analysed with generalized linear models using the software JMP, version 6.01. First, we analysed the associations between local species richness (separately for 1994 and 2004) and amount of low-intensity land-use (i.e. total proportion of arable land that was covered by set-aside fields, leys, cultivated pasture and short-rotation coppice), the three different heterogeneity indices and amount of forest. The two-way interaction term between amount of low-intensity land-use and amount of forest was also included in analyses of species richness in 1994 and 2004. We used generalized linear models with Poisson distribution and a log link function.

Second, we analysed the effect of all independent variables on changes in species richness between 1994 and 2004. Previous