Biodiversity Conservation in Agricultural Landscapes

Linking Farmers and Agri-Environmental Measures to Farmland Birds

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Abstract

Agricultural industrialization alters rural landscapes in Europe, causing large-scale and rapid loss of important biodiversity. The principal instruments to protect farmland biodiversity are various agri-environmental measures (AEMs) in the EU Common Agricultural Policy (CAP). However, growing awareness of shortcomings to CAP biodiversity integration prompts examination of causes and potential solutions. This thesis assesses the importance of structural heterogeneity of crop and non-crop habitats and evaluates some related aspects of the CAP for 2015-2020. This includes studies of crop diversification, organic farming and buffer strips, and their potential for supporting deteriorating farmland bird diversity in a forest-farmland gradient. It also evaluates the role of collaborative conservation, with particular attention to the Swedish Volunteer & Farmer Alliance (SVFA), as a tool for influencing farmers' engagement in AEMs as well as unsubsidized conservation.

Structural crop diversity, rather than the number of crop types in itself, positively affected farmland birds, especially in arable-dominated landscapes. Still, as almost all farms already met the CAP requirements for crop diversification, this policy may miss an important opportunity to deliver biodiversity benefits by setting limits too low and by neglecting structural crop diversity. The establishment of buffer strips along ditches boosted Skylarks and invertebrate numbers in adjacent cereal fields, while organic farming had only small and mixed effects on farmland birds, with both positive and negative effects on field nesters in the most arable-dominated landscapes and more forest-dominated landscapes, respectively. In general, landscape composition had a major effect on species richness, with different habitat preferences among field-nesting and non-crop-nesting birds. Social factors were more important for farmers' engagement in AEMs than for unsubsidized conservation, suggesting that production-impeding AEMs may have poor chances of acceptance in regions with prevailing productivist norms. We also found that SVFA promoted both AEMs and unsubsidized conservation, and that measures positively affected farmland bird diversity in the most arable-dominated landscapes. However, low implementation rates of measures across SVFA limited the large-scale impact, highlighting the importance of following up stakeholders' involvement.

This thesis suggests that farmland biodiversity conservation partly relies on policies that increase the structural heterogeneity of arable landscapes (*e.g.*, through crop diversification and establishment of buffer strips). This is especially important in regions where arable farming is predominant and farmland heterogeneity is low. We conclude the future of AEMs for biodiversity protection partly lies in better integration into cultures of farming communities, possibly through volunteer-based approaches as an alternative to centralized solutions.

Keywords: agri-environmental schemes, attitudes, buffer strips, collaborative conservation, crop diversity, landscape heterogeneity, non-crop habitat, organic farming, unsubsidized conservation, volunteer-based

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”It may be that when we no longer know which way to go that we have come to our real journey. The mind that is not baffled is not employed. The impeded stream is the one that sings.”

Wendell Berry

Contents

List of Publications

This thesis is based on the work contained in the following papers, referred to by Roman numerals in the text:

I. **Josefsson, J.**, Berg, Å., Hiron, M., Pärt, T. & Eggers, S. Does the CAP fit? Sensitivity of the farmland bird community to crop diversification in Sweden (manuscript).

II. **Josefsson, J.**, Berg, Å., Hiron, M., Pärt, T. & Eggers, S. (2013). Grass buffer strips benefit invertebrate and breeding skylark numbers in a heterogeneous agricultural landscape. *Agriculture, Ecosystems & Environment*, 181, 101-107.

III. **Josefsson, J.**, Lokhorst, A. M., Pärt, T., Berg, Å. & Eggers, E. The role of collaborative biodiversity management in facilitating farmers' intentions to adopt subsidized and unsubsidized conservation measures (manuscript).

IV. **Josefsson, J.**, Pärt, T., Berg, Å., Lokhorst, A. M. & Eggers, S. Farm-scale implementation and ecological effects of bird conservation measures in response to the Swedish Volunteer & Farmer Alliance (manuscript).

The contribution of Jonas Josefsson to the papers included in this thesis was as follows:

I. Main author, data management and analysis. Study design with Sönke Eggers, Åke Berg and Tomas Pärt.

II. Main author, data management and analysis. Did most of the writing with significant contribution by Sönke Eggers. Study design and field work with Sönke Eggers.

III. Main author, data management and analysis. Study design with Sönke Eggers, Åke Berg, Anne Marike Lokhorst and Tomas Pärt.

IV. Main author, data management and analysis. Study design with Sönke Eggers, Åke Berg and Tomas Pärt.

1 Introduction

1.1 The industrialization of agriculture

The story of agriculture in Europe over the last half-century follows the trajectory of society in general and is largely about industrialization. To be sure, agricultural industrialization has had short-term gains: labor savings and higher productivity in terms of per-hectare outputs (Tilman et al., 2002). When told from a wider perspective, however, the social and ecological sustainability of recent and ongoing changes to agricultural practices is questionable.

The rural landscape can be read through the signs of this industrialization. Smaller farms have, and continue, to capitulate under economic pressures and become incorporated in businesses that gradually grow larger. To cope with increased economic pressures, fields have been enlarged to achieve greater machine and operator efficiency, at the expense of field boundaries, hedges and other marginal elements (Stoate et al., 2001; 2009; Figure 1). External inputs of mineral fertilizers sustain soil fertility without the use of green (*i.e.*, nitrogen-fixating crops) or animal manure. This decoupling of farming practices from the cycling of nutrients, together with changing economic conditions, has impelled a specialization of farm businesses (Wretenberg et al., 2007). In this way, many farms in regions with favorable conditions cultivate only a handful of high-yielding crop varieties while farms in regions where conditions limit the potential for intensification instead focus on ley cultivation and animal husbandry (Figure 1). Due to the low profitability of livestock farming, these latter regions often suffer from farm abandonment leading to loss of farmland through reforestation or natural succession into shrub lands (Wretenberg et al., 2007). Another significant component of agricultural intensification is the introduction of and increase in the use of pesticides, upheld by their short-term profitability, although the long-term sustainability of their use can be questioned (Geiger et al., 2010; Goulson, 2013).

Thus, rural landscapes have transformed at two scales – at the field scale through increased inputs and outputs, and at the landscape scale through landscape simplification of both crop and non-crop areas, resulting in loss of heterogeneity. Furthermore, there are two simultaneous, but spatially differentiated processes at the landscape scale: one of intensification in regions with large areas of high soil fertility, where farms become fewer, larger and more intensively managed, and one of land abandonment in regions with poorer conditions for farming.

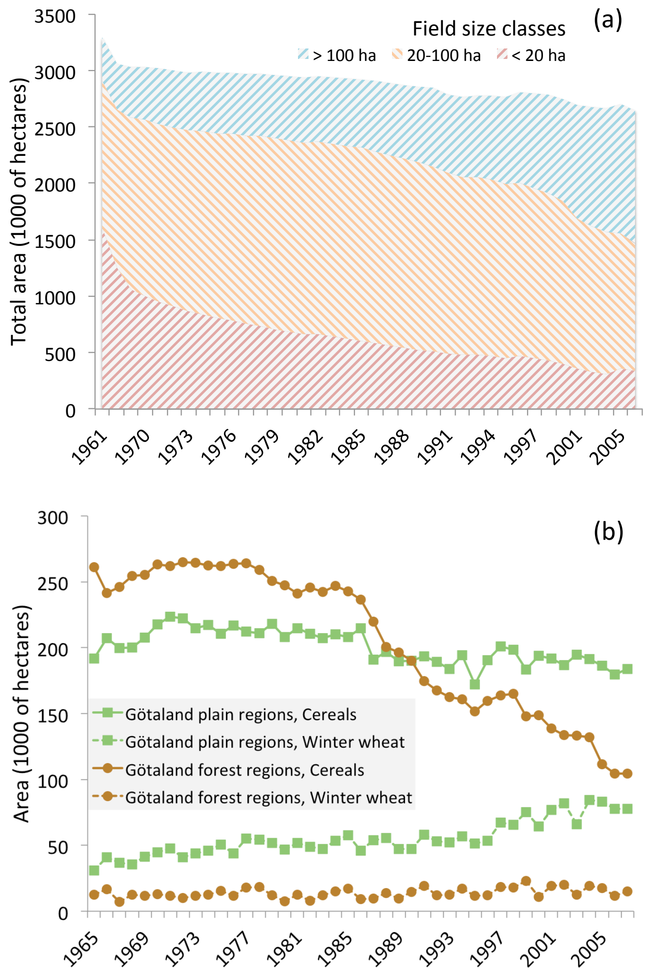


Figure 1. a) The size of agricultural fields in Sweden between 1961 and 2007. Fields have gradually changed over time from smaller to larger. b) Area of cereals and winter-sown wheat in two contrasting regions in Sweden. The area cultivated with cereals has decreased in forest regions (Götaland forest regions), while they have remained constant in landscapes more dominated by arable farming (Götaland plain regions). In these regions, the proportion winter-sown wheat has increased. Data from the Swedish Board of Agriculture.

1.2 Biodiversity in agricultural landscapes of Europe

In Europe, nearly half of all land is under some form of agricultural use, and much of the continent's wildlife is tied to, and thus affected by, farming (Stoate et al., 2009; Kleijn et al., 2011). From a historical perspective, agriculture created a varied farmscape, where arable fields and meadows, together with interstitial elements such as grass margins, paths, temporary water pools, and stone fences held a large diversity of species. This biodiversity has supported functions important to agriculture, including pest control, crop pollination, and sustainment of soil fertility (Bianchi et al., 2006; Wall & Nielsen, 2012; Kennedy et al., 2013).

1.2.1 Agricultural intensification and biodiversity

However, there is much evidence that biodiversity fares badly from the intensification of agricultural practices, outlined in the previous section (*e.g.*, Pimentel et al., 1992; Krebs et al., 1999). While troublesome in its own right, wide-scale biodiversity loss may also threaten the delivery of important ecosystem functions (Karp et al., 2013; Kennedy et al., 2013). Continuous application of agro-chemicals might partly conceal the deterioration of biological pest control and soil services at present, but future shortages of phosphorus supplies (Cordell & White, 2011), or increased knowledge of detrimental effects of modern pesticides (Goulson, 2013), might render these functions very important in the long-term.

Biodiversity is a very broad concept, including genetic diversity, the diversity of organisms and also diversity of ecosystems – whether in a specific area, biome, or across our planet. Apart from possible intrinsic values (value in itself, whether subjective or objective) of biodiversity, and direct instrumental values (*e.g.*, pleasure or aesthetics), different aspects of biodiversity also underpin many functions that are central for human health and livelihoods (Millenium Ecosystem Assessment, 2005). However, whether one chooses to define its value or not, numerous national objectives as well as international agreements for the protection of biodiversity undeniably indicate that biodiversity is highly valued in our society (Maes et al., 2012; Larsen, 2015). While several aspects of biodiversity are possibly affected by agricultural intensification, and thus hold significance for the questions addressed in this thesis, I concentrate on taxonomic diversity (*i.e.*, species diversity). Taxonomic diversity comprehends a large proportion of the variability in biodiversity, and is also possible to study over large spatial and temporal scales.

Aside from the direct effects on biodiversity, modern farming practices also pose important environmental threats beyond agricultural landscapes. For example, nutrient surpluses in areas where agricultural runoff converges, such as sensitive coastal areas, often result in ecological changes to these ecosystems (Ulen et al., 2007). Also, production and application of mineral fertilizer as well as livestock farming increase agriculture's carbon footprint through greenhouse-gas emissions (McMichael et al., 2007).

1.2.2 Farmland bird declines

Studies at national and continental scales show steep population declines of farmland birds across Europe, particularly in countries with a longer history of agricultural intensification (Chamberlain et al., 2000; Donald et al., 2001; Wretenberg et al., 2006). While the role of birds in agri-ecosystems is somewhat elusive, their importance as providers of pest control and other functions is increasingly acknowledged (Şekercioğlu et al., 2004; Şekercioğlu, 2006; Karp et al., 2013). Farmland bird species vary in life history traits such as foraging and nesting behavior, where some species nest and forage in agricultural fields, while others rely solely on non-crop elements for nesting but forage in agricultural fields (Hiron et al., 2015). They also display a spectrum of diets, ranging from exclusively seed-based to exclusively invertebrate-based diets (Holland et al., 2006). Nevertheless, these species share a dependency on habitats and conditions created by agricultural land-use practices.

At the field scale, declines in farmland bird populations have been attributed mainly to reductions in food supplies due to the increased use of agro-chemicals (Potts 1986; Boatman et al., 2004), and increased shares of autumn-sown cereals and fast-growing crop varieties whose dense swards limit food accessibility (Atkinson et al., 2005; Eggers et al., 2011). Also, faster growth of grass has led to earlier and more frequent cutting that causes high rates of nest destruction for species nesting in grassland habitats (Newton, 2004; Perlut et al., 2006). At a larger spatial scale, the loss of field boundaries, ditches and other non-crop elements, together with the regional specialization of farming businesses (in contrast to diversified agricultural production), has gradually dissolved the earlier heterogeneous farmland landscape (Benton et al., 2003). Consequently, these landscapes have progressively deteriorated from meeting the varied requirements of the farmland bird community.

1.3 Biodiversity integration in the EU Common Agricultural Policy

The integration of biodiversity aspects in the EU Common Agricultural Policy (CAP) began with the 1992 MacSharry[1] reform, which made agri-environmental measures (AEMs) compulsory for Member States (EEC No. 2078/92). This integration further developed through reforms in the following decade. Devised by each Member State, AEMs aim to encourage farmers, through payments for costs or income forgone, to protect and enhance ecological conditions on their farmland for the benefit of biodiversity and other environmental values. Farmers voluntarily choose whether to participate or not in various appointed AEMs, as well as the scope and location of these measures. With the last decade's CAP reforms, however, a narrative centered on the necessity for higher production within the EU has returned, an idea that contradicts earlier judgments of reaching a state of environmental welfare through production limitation. Although this recently revived description is doubtful (Tilman et al., 2011), together with the necessity to address issues of climate change and bioenergy, it has still pushed biodiversity conservation down the priority ladder. Thus, instead of further advancing biodiversity protection in the CAP, there are vague objectives on ecosystem services, which have been criticized for having little prospect of realization (Kleijn et al., 2014; Melathopoulos et al., 2015).

The current CAP period, running from 2015 to 2020 (EU No. 1307/2013), has four main instruments to develop agriculture into an ecologically sustainable sector, namely farm-scale crop diversification, permanent grassland retention, ecological focus areas (non-crop elements and extensively managed crops), and organic farming. Organic farming is promoted through AEM payments while the other three instruments are included as "greening" components in the CAP and are compulsory for farms to receive the Single Farm Payment (subsidies to farmers on a per-hectare basis and decoupled from production). Thus, fund allocation directed towards AEMs has decreased considerably (European Parliament, 2013), which has resulted in a contentious debate regarding the lack of powerful tools for biodiversity protection in the latest CAP reform (Dicks et al., 2014; Peer et al., 2014).

1. Ray MacSharry was the European Commissioner of Agriculture at the time. He aimed to reorient the original production-oriented objectives of the CAP to a more multifunctional view of farming. The CAP has since undergone successive change through Agenda 2000, the 2003 reform, the 2008 Health Check and most recently the CAP Post-2013 reform.

1.4 Agri-environmental measures and biodiversity

Since the seminal paper by Kleijn & Sutherland (2003) quantification of effects from AEMs on biodiversity has been an on-going endeavour, but it also prompted extensive research aimed at improving their effectiveness. The current position is that AEMs can have moderate local effects on biodiversity depending on the type of AEM and the taxon studied. Generally, AEMs directed at areas outside production (such as field margins or hedgerows) are suggested to deliver greater benefits than AEMs aimed at productive areas such as arable fields or grasslands (Batáry et al., 2015), and plants are suggested to benefit more than more mobile taxa such as birds (Batáry et al., 2011). However, several key constraints in the implementation of AEMs have contributed to a policy that has failed to protect European farmland biodiversity to any great extent (Kleijn & Sutherland, 2003; Whittingham, 2006).

1.4.1 Landscape moderation

Firstly, biodiversity outcomes of AEMs may be moderated by the surrounding landscape, with noticeable effects only in structurally simple and intensively farmed landscapes, but with only marginal effects in more complex landscapes (Batáry et al., 2011; Concepción et al., 2012). In these landscapes, extensive farming and high amounts of non-crop habitats already support a relatively high level of biodiversity (Tscharntke et al., 2005; Kleijn et al., 2011). Yet, there is no landscape- or situation-based approach to the implementation of most AEMs, and current economic incentives have not been sufficient to attract farmers in intensively managed and high-yielding arable regions (Kleijn & Sutherland, 2003; Quillérou & Fraser, 2010). Instead, participation in AEMs has concentrated to extensively farmed regions where their efficiency is hampered, while intensive farming has largely stayed on the path of business as usual. For instance, organic farming – one of the most commonly and widely implemented AEMs – is in Sweden largely concentrated in forest-dominated regions where arable farming is only marginal (Official Agricultural Statistics, Swedish Board of Agriculture). Thus, it is questionable whether these subsidies are targeting areas where they have the largest impact on biodiversity (Winqvist et al., 2012; Tuck et al., 2014).

1.4.2 Farmers' participation

A second shortcoming of AEM implementation, the top-down method of their design and implementation, has formed an agri-environmental policy that is not rooted in farming culture (Burton et al., 2008; Burton & Paragahawewa, 2011). This has been suggested to result in weak intrinsic motivations behind AEM participation among farmers (Lokhorst et al., 2011). Consequently, AEMs currently sustained by the AEM payments risk being abandoned if/when subsidies disappear (Herzon & Mikk, 2007). The negative dispositions among farmers towards AEMs has also led to a resistance against options deemed too demanding in terms of management effort (Butler et al., 2010) and ultimately in prevailing productivist ideals among many farmers (Ahnström et al., 2009; de Snoo et al., 2012). This affects uptake of AEMs, and thus their effectiveness, negatively (Kleijn & Sutherland, 2003). Therefore, it is an important task to find ways to increase acceptance for conservation measures, especially in regions of intensive agriculture.

1.5 Collaborative approaches to farmland biodiversity conservation

Growing awareness of the shortcomings of the "one-size-fits-all" approach to farmland biodiversity conservation has incited some Member States, including the UK and the Netherlands, to instigate collective AEM applications for farmers in order to increase landscape-wide implementation (Franks & Emery, 2013; van Dijk et al., 2015). In addition to such policy-assisted approaches, biodiversity conservation is increasingly tackled also through various volunteer-based collaborative efforts (Miller et al., 2011). If underfunding of biodiversity conservation in the CAP continues, such projects are likely to be essential to farmland biodiversity protection in the near future. However, few collaborative projects evaluate their outputs (*e.g.*, number of interventions), and even fewer assess impacts on biodiversity (Koontz & Thomas, 2005; but see *e.g.*, Santangeli et al., 2015). This is likely a result of limited project funds, which sets quantification of such parameters aside.

1.5.1 The Swedish Volunteer & Farmer Alliance

The Swedish Volunteer & Farmer Alliance (SVFA) was initiated by BirdLife Sweden and the Rural Economy and Agricultural Societies, and has engaged farmers, mainly in plain regions, to adopt conservation measures for farmland birds (Eggers & Engström, 2007). SVFA is modelled after RSPB's Volunteer and & Farmer Alliance in the UK (Smallshire et al., 2004). Almost 300 farmers across Sweden's main agricultural regions participated in SVFA, whose framework comprised farm-scale bird inventories by more than 200 birdwatchers from BirdLife Sweden's network, consultative visits, and follow-up inventories to allow evaluation of bird-population responses to implemented actions (Figure 2). SVFA engaged farmers with conservation advisors from the Rural Economy and Agricultural Societies in face-to-face consultations focused on farm-tailored advice on AEMs and unsubsidized conservation measures to improve conditions for farmland birds (Table 1).

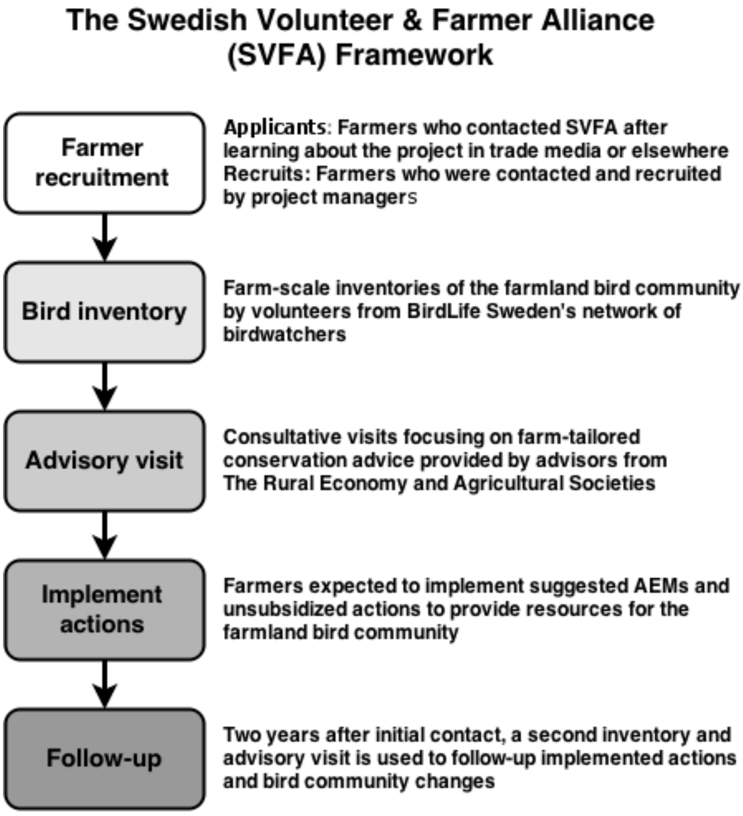


Figure 2. Schematic view of the collaborative framework used in the Swedish Volunteer & Farmer Alliance.

Table 1. *List of AEMs in the Swedish Rural Development Program 2007-2013 and the unsubsidized conservation measures promoted within the Swedish Volunteer & Farmer Alliance (SVFA).*

Measure	Description
AEMs in the Swedish Rural Development Program 2007-2013† *promoted within SVFA*	
Pasture management	Payments to maintain well-grazed, permanent open grasslands without successional plant species
Elements with nature or cultural values	Payments to maintain landscape elements with nature or cultural value (*e.g.*, cairns)
Buffer strips	Payments to create grass strips to minimize agricultural run-off into waterways
Wetlands	Payments to manage wetlands to improve existing wetlands.
Conservation headlands	Payments for selective application of pesticides in arable field borders.
Unsubsidized conservation measures promoted within SVFA	
Winter feeding	Winter feeding of birds or allowing access to grain stock rooms
Game and pollinator habitat	Cultivating cover crops or strips of flowering plants
Cultivating extensively managed crops	More spring-sown crops, Salix, set-asides, grass for seed, oil crops
Planting bushes and trees	Planting and managing hedgerows, bushes and solitary trees
Bare patches	Create bare, uncultivated patches and leave existing bare patches from water-logging over the winter
Unthreshed patches	Save unthreshed patches and/or strips of cereal and clover crops
Nest boxes	Erecting nest boxes at farmsteads and in environments such as woodlands and gardens
Bird-adapted field management	Timing of field management activities (fertilizer and pesticide applications, harrowing and harvesting) to ensure chick survival
Embankments	Creating non-crop habitat by establishing in-field embankment strips
In-field islands	Managing in-field islands, *e.g.*, by clearing overgrown vegetation
Unmanaged patches	Keeping areas with weeds

†AEM descriptions are adapted from Hiron et al., 2013a.

1.6 Knowledge gaps

1.6.1 Importance of crop and non-crop heterogeneity for biodiversity

Previous studies indicate that effective farmland biodiversity protection may rely on policies that increase the number (compositional heterogeneity) and/or spatial arrangement (configurational heterogeneity) of both crop and non-crop habitats at multiple spatial scales (*e.g.*, Benton et al., 2003; Billeter et al., 2008). Farmland biodiversity is expected to benefit from increased heterogeneity through habitat complementation (for species needing several habitats) and niche differentiation effects (for species with different habitat requirements), supporting more species, but also a higher abundance (Fahrig et al., 2011).

Increased crop diversification

Crop diversification is potentially a viable biodiversity conservation strategy that increase heterogeneity of agricultural landscapes without requiring that agricultural land be taken out of production (Khoury et al., 2014), and might target species of high conservation concern that benefit from structural heterogeneity (Bastolla et al., 2005; Butler et al., 2007). However, it is still unclear to what extent crop heterogeneity of arable fields contributes to biodiversity in different landscapes (but see Hiron et al., 2015; Gil-Tena et al., 2015). This lack of knowledge is due to a scarcity of detailed, but wide-scale, studies that disentangle effects of crop and non-crop cover types and their spatial arrangement on biodiversity. Such studies are complicated partly by the often-strong correlation between compositional and configurational components of heterogeneity, which obscures separation of their effects (Fahrig et al., 2011; Hiron et al., 2015).

Understanding the sensitivity of biodiversity to crop diversification also requires to explicitly link expected functions of different crop types to the habitat requirements of the species or species group of interest (*e.g.*, availability of food supplies and nesting sites in the case of farmland birds; Eggers et al., 2011; Vasseur et al., 2012). Despite this lack of background information, crop diversification regulations are already imposed in the CAP. Hence, it is important to assess whether crop diversification in the CAP is adequately designed to reduce the adverse effects of intensive farming on biodiversity.

Increasing structural heterogeneity using subsidized buffer strips

A commonly adopted AEM in intensive farmland is the establishment and maintenance of grass buffer strips on cereal field edges (Figure 3). These strips are mainly used to reduce erosion and agro-chemical runoff into surface water.

Between 2006 and 2012, buffer strips covered only between 5 and 11,000 hectares (ha) of Swedish arable land (Official Agricultural Statistics, the Swedish Board of Agriculture), while the potential has been estimated at 100,000 ha (Rabinowicz, 2010). Often assumed to also provide refuge and food for invertebrates, small mammals and birds (Marshall & Moonen, 2002), buffer strips thus represent a potentially important conservation tool to target diversity loss in arable-dominated landscapes. Evidence for this idea is scarce, however, and it remains unclear if densely vegetated buffer strips benefit biodiversity in the structurally complex landscapes of Northern Europe.

Figure 3. A spring-sown field with a densely vegetated buffer strip in Lagga, Uppland.

1.6.2 Impacts of collaborative efforts

As stated in the introduction, recent CAP reforms have diminished subsidies for biodiversity protection (Dicks et al., 2014; Peer et al., 2014). As a consequence, few AEMs currently target farmland heterogeneity loss, despite several studies recognizing this aspect of agricultural intensification as a key driver of farmland biodiversity declines (Benton et al., 2003). While measures with a relatively strong evidence base regarding their benefits to farmland birds are available (reviewed in Williams et al., 2013), several are not subsidized as AEMs, undoubtedly limiting their uptake on farms. Here, collaborative efforts such as the Swedish and British Volunteer & Farmer Alliances aim to engage farmers in such unsubsidized measures, alongside increasing uptake of AEMs, to reverse negative farmland bird trends. However, systematic evaluations of this form of collaborative projects are needed, not only to determine their efficiency as funding towards biodiversity conservation is limited, but also to identify areas where such methods might be improved (Koontz & Thomas, 2005; Lubell, 2004).

2 Aims

The overarching goals of this thesis are to i) explore how farmland bird diversity relates to farmland heterogeneity and agri-environmental measures (AEMs) in landscapes with different proportions of arable land across southern Sweden and ii) identify ways to improve farmland biodiversity conservation with particular attention to the role of collaborative conservation in the Swedish Volunteer & Farmer Alliance.

The specific aims were to:

- Identify the relative importance of components of structural heterogeneity of crop fields and non-crop habitats for farmland bird populations (Paper **I**)

- Evaluate some key aspects of current agri-environmental policy, including crop diversification, organic farming and buffer strips on cereal field edges, regarding their capability to support biodiversity (Papers **I & II**)

- Examine how the collaborative approach of the Swedish Volunteer & Farmer Alliance influenced farmers' engagement in subsidized AEMs and unsubsidized conservation measures (Paper **III**)

- Evaluate the effectiveness of the Swedish Volunteer & Farmer Alliance in moderating farmland bird declines in different landscapes, from forest-dominated to arable-dominated landscapes (Paper **IV**)

3 Methods

3.1 Farm and field selection

Data for Papers **I**, **III** and **IV** originated from the 295 farms, 201 conventional and 94 organic, participating in the Swedish Volunteer & Farmer Alliance (SVFA). These farms covered all main agricultural areas in Sweden, spanning roughly 130,000 km^2 across the six southern production regions (Sveriges nationalatlas: Jordbruket 1992; relatively homogeneous areas regarding characteristics such as climate, topography and soil structure, see Figure 4). For various reasons (unavailable land-use data, lack of reinventories and survey responses), the number of farms varied across studies. Papers **I**, **III** and **IV** used samples of 178, 139 and 103 farms, respectively. Paper **III** also included a randomly selected group of farmers (n = 299) acquired from Statistics Sweden, covering the same geographical range and stratification as the SVFA.

Field work for Paper **II** was carried out in Uppsala county in the south-central Swedish plain (59°40′ N; 17°15′ E), where the landscape is dominated by crop fields interspersed with forests, small areas of semi-natural grasslands and wetlands. Twenty-four cereal fields with and without buffer strips were matched pairwise across multiple criteria to account for potentially confounding effects of sowing regime (spring/autumn-sown), field size, ditch size and other landscape elements affecting Skylark *Alauda arvensis* breeding numbers and invertebrate abundance.

3.2 Data collection and methodology

3.2.1 Farmland bird inventories

Papers **I** and **IV** used farm-scale inventories from farms in SVFA, of nesting and/or territorial individuals of a subset of 29 typical farmland bird species (*cf.*, Bibby et al., 1992). We distinguished between two trait-based groups: species that both nest and forage in fields (field-nesting species), and species that forage both within agricultural fields and non-crop habitat, but rely solely on non-crop elements for nesting (non-crop-nesting species). These two groups are expected to respond differently to heterogeneity of crop and non-crop habitats and might also react differently to conservation measures.

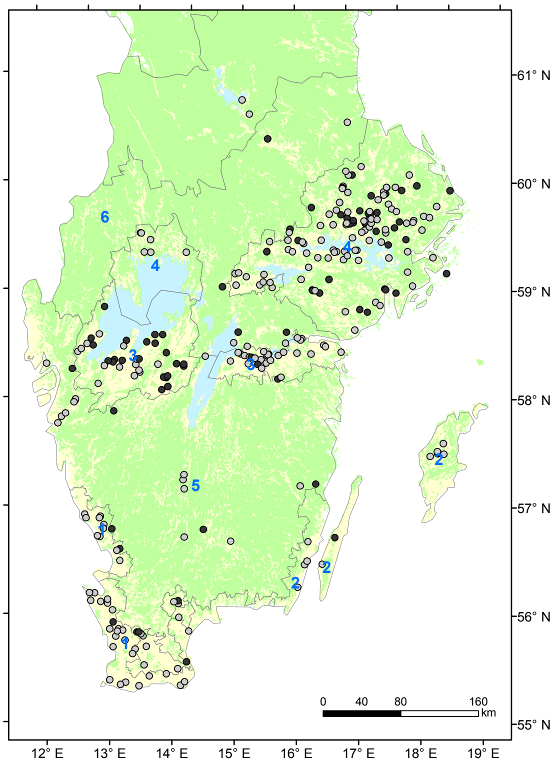


Figure 4. Map of Sweden, showing the location of conventional (gray) and organic (black) farms within the Swedish Volunteer & Farmer Alliance. Production regions: Götaland's southern (1) and northern (2) plains and Svealand's plains (4), the regions with most farmland and large farms, specializing in production of cereals; Götaland's mosaics (3), mosaic landscapes with mixed forest and farmland, more cattle farming than in the plain regions; Götaland's (5) and Svealand's (6) forested regions, dominated by forest with smaller areas of farmland interspersed in the forest landscape, small farms with less intensive production, often with cattle farming and semi-natural pastures.

The group of field-nesting species (n = 10) included: Grey Partridge *Perdix perdix*, Common Quail *Coturnix coturnix*, Corn Crake *Crex Crex*, Common Pheasant *Phasianus colchicus*, Northern Lapwing *Vanellus vanellus*, Eurasian Curlew *Numenius arquata*, Eurasian Skylark *A. arvensis*, Meadow Pipit *Anthus pratensis*, Western Yellow Wagtail *Motacilla flava*, Corn Bunting *Emberiza calandra*.

Non-crop-nesting species (n = 19) included: Montagu's Harrier *Circus pygargus*, Common Kestrel *Falco tinnunculus*, Common Snipe *Gallinago gallinago*, Barn Swallow *Hirundo rustica*, Common House Martin *Delichon urbicum*, White Wagtail *Motacilla alba*, Thrush Nightingale *Luscinia luscinia*, Northern Wheatear *Oenanthe oenanthe*, Whinchat *Saxicola rubetra*, Common Grasshopper Warbler *Locustella naevia*, Common Whitethroat *Sylvia* *communis*, Red-backed Shrike *Lanius collurio*, Common Starling *Sturnus vulgaris*, House Sparrow *Passer domesticus*, Eurasian Tree Sparrow *Passer montanus*, Common Linnet *Carduelis cannabina*, European Goldfinch *Carduelis carduelis*, Ortolan Bunting *Emberiza hortulana*, Yellowhammer *Emberiza citrinella*.

3.2.2 Heterogeneity and landscape variables

To discern effects of farm-scale structural crop diversity, compared to crop diversity without distinction of crop structure, we derived two crop diversity metrics. These were Shannon's diversity indices from the relative proportions of (i) crops without distinction of crop structure or function ($H'_{indiscriminate}$), and (ii) crops aggregated in eight structural crop classes according to their structure and management, thus being functionally different in providing resources for farmland birds ($H'_{structural}$; see Table 2 and Figure 5). To compare their relative impact on farmland birds, $H'_{indiscriminate}$ and $H'_{structural}$ were assessed in two separate models, all other terms being the same. We also included the mean field size of the inventoried farm areas as a measure of configurational crop heterogeneity.

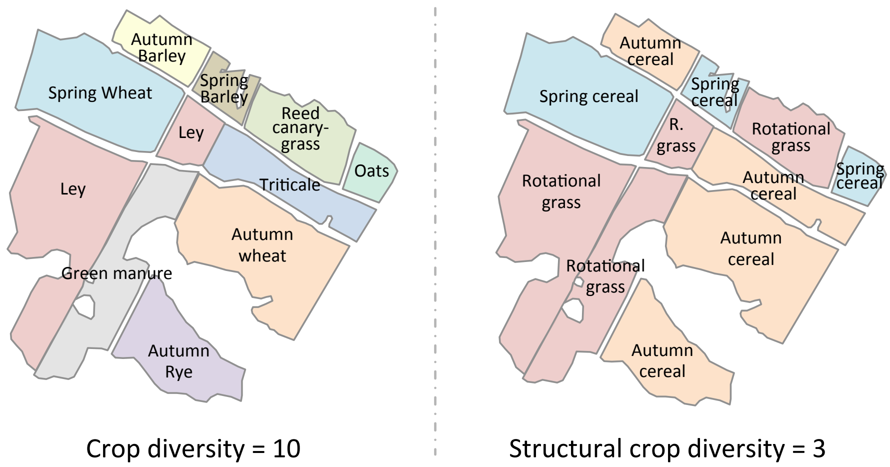


Figure 5. Illustration of the resulting number of crops using the two approaches to quantify crop diversity: without distinction of crop structure, or differentiating crops according to their structure and management (structural crop diversity; see Table 2).

Table 2. *Structural crop classes based on vegetation structure and management used to calculate structural crop diversity ($H'_{structural}$). Data was summarized from the Swedish Board of Agriculture's crop database.*

Structural crop class	Crop types
Autumn-sown cereals	Wheat†, Triticale, Rye, Barley†
Spring-sown cereals	Barley†, Oats, Wheat†, Maslin, Green fodder cereal
Other autumn-sown crops	Rapeseed†, Turnip rape†
Other spring-sown crops	Rapeseed†, Peas, Sugar beet, Broad bean, Potato, Flax, Vegetable cultivation, Corn, Green fodder, Brown bean, Turnip rape†, Oil radish, Sunflower, Hemp
Rotational grass	Ley, Green manure, Grass for seed production, Reed canary-grass
Permanent grassland	Pasture, Wetland, Hayfield, Forest pasture
Extensive cultivation	Set-aside, Forage for game, Diversity set-aside, Unused arable land
Perennial bushes	Willow *Salix spp.*, Christmas tree plantation, Fruit and berry cultivation, Other horticulture

†Crops that exist as both autumn- and spring-sown varieties

To evaluate effects of non-crop elements, we included (i) the proportion of non-crop area (excluding forest) and we also specifically accounted for two structural landscape elements that influence farmland bird numbers: (ii) trees and shrubs, and (iii) in-field islands, and categorized farms according to farming system (conventional or organic; Figure 6). Finally, to assess direct effects of landscape composition (from forest- to arable-dominated) on farmland birds, as well as landscape-moderation of effects of crop heterogeneity and agri-environmental measures, we used the proportion of arable land in a 1000-meter radius around the inventoried farm areas (Figure 6).

3.2.3 Biodiversity in fields with and without buffer strips

Skylarks *A. arvensis* were counted over five visits at the 24 paired study fields in intervals of one week between May 22nd and June 21st. Study plots extended into fields as an arc with a 100-meter radius (1.57 ha). Further, we placed three pitfall traps in each field: in the field border, and at 15 and 30 meters into the field. Traps were set at the date of the first skylark count (May 22nd) and were emptied concurrent with skylark counts. From the samples, we counted the number of beetle *Coleoptera* and spider *Arachnida* individuals larger than 0.5 cm (> 90 % of the spider sample). We focused on beetles and spiders since these two orders constitute the bulk of the diet of skylark chicks (Holland et al., 2006).

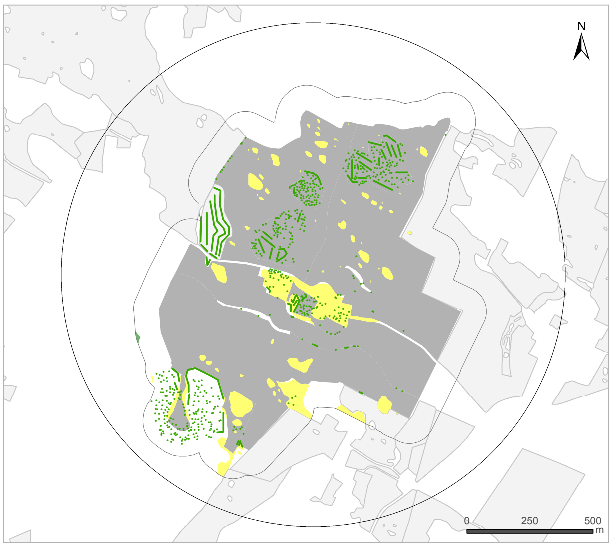


Figure 6. Schematic of a farm with digitized crop and non-crop habitats and elements. Dark gray, inventoried arable fields; Light gray, arable fields outside inventory area; Yellow, non-crop habitats and in-field islands; Green, trees and shrubs. 1000-m circle, area from which proportion of arable land was calculated.

3.2.4 Motivational differences between subsidized and unsubsidized conservation

Another of our aims was to examine if and how SVFA affected farmers' intentions to implement subsidized AEMs and unsubsidized conservation measures. Here, we used the theory of planned behavior (TPB; Figure 7) to understand how intentions, and underlying behavioral determinants, to adopt AEM and unsubsidized conservation measures differed between farmers in SVFA and a randomly selected group of farmers. The behavioral determinants in the TPB include: attitudes (a personal evaluation of whether the behavior is positive or negative), subjective norms (the perceived social pressure to engage in the behavior), and perceived behavioral control (the perceived practicability to perform the behavior). To examine the importance of conservationist norms and whether SVFA encouraged their manifestation, we augmented our model by including self-identity (the extent that conservation measures were part of farmers' self-identity) as an additional behavioral determinant. The effects of self-identity on intention were hypothesized to be both direct and indirect (*i.e.*, mediated through its effects on attitude, subjective norms and/or perceived behavioral control; Figure 7).

SVFA comprised two sub-groups of farmers according to way of recruitment: i) SVFA applicants, who contacted project managers after seeing SVFA advertisements in trade media, and ii) SVFA recruits, who were contacted randomly by project managers and presumably had a baseline interest for conservation comparable to the randomly selected group (see

Figure 2). Correspondingly, differences between the randomly selected group and SVFA applicants only, and not between the random group and SVFA recruits could potentially result from SVFA drawing farmers with an inherent interest in nature conservation, while differences also between the control and recruits would indicate changes attributed to SVFA. Explicitly, we addressed the following questions: (Q1) Do SVFA applicants and recruits differ from the control group regarding intention to engage in nature conservation? (Q2) Are between-group differences in intention explained by corresponding differences in behavioral constructs (attitudes towards the conservation actions, subjective norm (perceived social pressure), perceived behavioral control and the conservationist self-identity)? (Q3) What are the relative strengths of direct and indirect effects of self-identity on intention (*i.e.*, mediation through attitude, subjective norms and/or perceived behavioral control)?

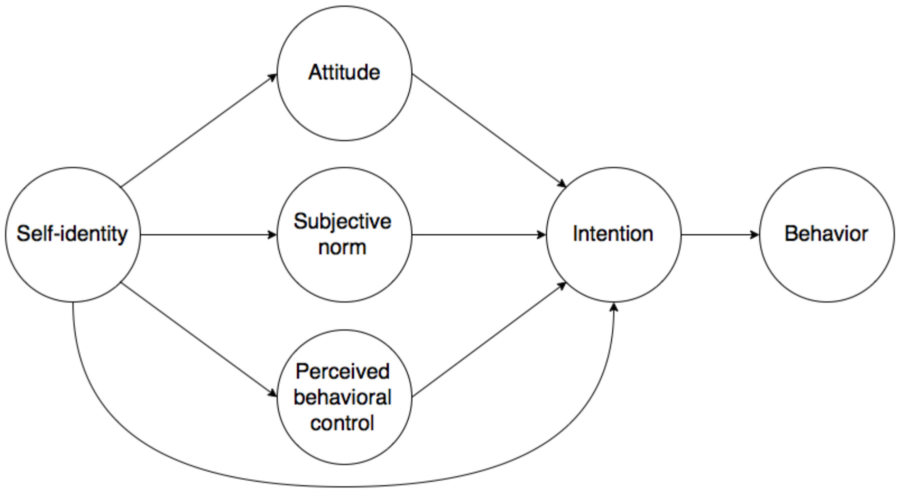


Figure 7. The theory of planned behavior extended to incorporate the influence of self-identity. Effects of self-identity on intention could be both direct and indirect through its influence on attitude, subjective norm and perceived behavioral control (*i.e.*, mediated effects).

3.2.5 Impacts of the Swedish Volunteer & Farmer Alliance

To identify potential effects of conservation measures implemented within SVFA on farmland bird populations, we asked i) to what extent farms in SVFA implemented conservation measures and ii) whether these measures impacted the species richness and abundance of farmland birds between baseline-inventories before and re-inventories after measures were implemented. We simultaneously accounted for changes in structural crop diversity (see Section 3.2.2) between inventories, and evaluated how overall population trends and potential effects of conservation measures were affected by the composition of the landscape that farms were situated in, from forest-dominated to arable-dominated landscapes.

3.3 Statistics

In Papers **I**, **II** and **IV**, given the nature of the data (species richness and abundance counts and nested study designs), we used Poisson generalized linear mixed models (GLMMs) to analyze species richness and abundance of farmland birds. Models also included an observation level random effect to reduce parameter estimate bias from overdispersion when needed (Bolker et al., 2009). To account for uncertainty in the model selection process we ran all subsets of the global model, calculated the Akaike Information Criterion with a correction for finite sample sizes (AICc) and ΔAICc and subsequently derived relative variable importance (RVI) and model averaged parameter estimates using models with $\Delta AIC_c < 4$ and their relative AICc weights (Burnham & Anderson 2002).

In Paper **III**, we used structural equation modeling (SEM) to estimate the influence of behavioral determinants on intention to implement subsidized/unsubsidized conservation measures. The resulting pathway coefficients can be viewed as analogues to regression coefficients and indicate the influence of one construct on another. Thus, when formulating the model, hypothesized relationships between constructs were estimated, while those pathways that were hypothesized to have no relationship were fixed at zero.

4 Results and discussion

Of the 29 bird species surveyed, the most commonly occurring species across SVFA farms between 2006 and 2013 were: Eurasian Skylark *A. arvensis* (occurring on 97 % of farms), Yellowhammer *E. citrinella* (85 % of farms), Common Whitethroat *S. communis* (85 % of farms), Whinchat *S. rubetra* (72 % of farms), Northern Lapwing *V. vanellus* (67 % of farms) and Common Starling *S. vulgaris* (59 % of farms; Figure 8).

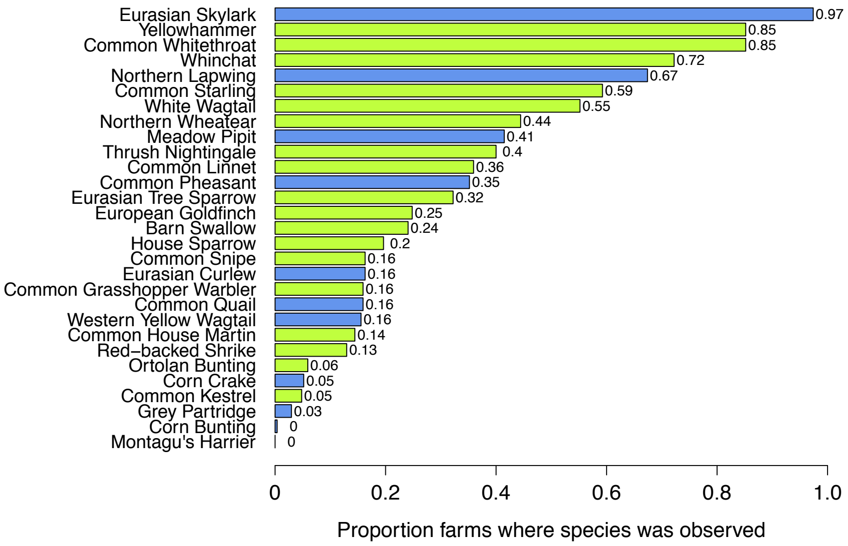


Figure 8. Species-specific occurrence (proportional presence) of field-nesting (blue bars) and non-crop-nesting farmland bird species (green bars) on SVFA farms (n = 270).

4.1 Landscape heterogeneity and farmland birds

4.1.1 Structural crop diversity

Papers **I** and **IV** highlight the importance of structural crop diversity ($H'_{structural}$; *cf.*, Figure 5) for farmland bird diversity, while crop diversity, without distinction of crop structure did not explain variation to the same extent (not shown here). In Paper **I**, the effect of structural crop diversity was mainly apparent for richness and abundance of non-crop nesters (Figure 9), and in Paper **IV**, the effects were instead apparent for abundance of field-nesting species (not shown here). For the species richness of non-crop nesters, the effect was mostly evident in landscapes dominated by arable land (Figures 9 and 10). These differences in observed effects of crop diversity could depend on for example model complexity (Paper **I** accounted for non-crop habitats and farming system, whilst Paper **IV** did not), which prompts further investigation of these inconsistencies. Nonetheless, these results underline the need for conservation strategies that properly address the heterogeneity of arable crop fields in these landscapes (Batáry et al., 2011; 2015).

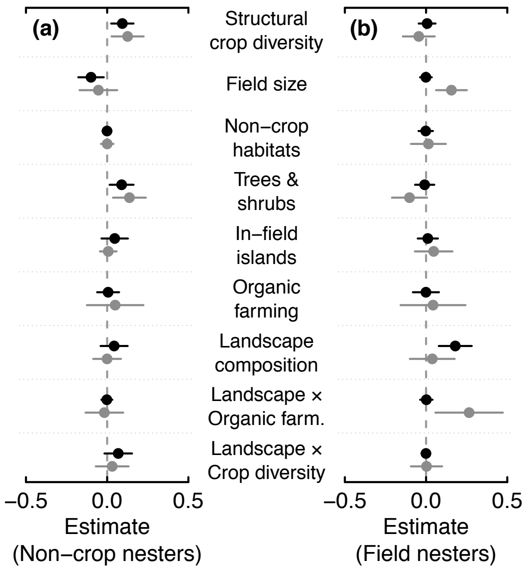


Figure 9. Selected results from GLMMs showing estimated effects and 95 % CI of structural heterogeneity, farming system (organic farming) and landscape composition (proportion of arable land within a 1000-meter radius) on species richness (black) and abundance (gray) of a) non-crop-nesting, and b) field-nesting farmland bird species.

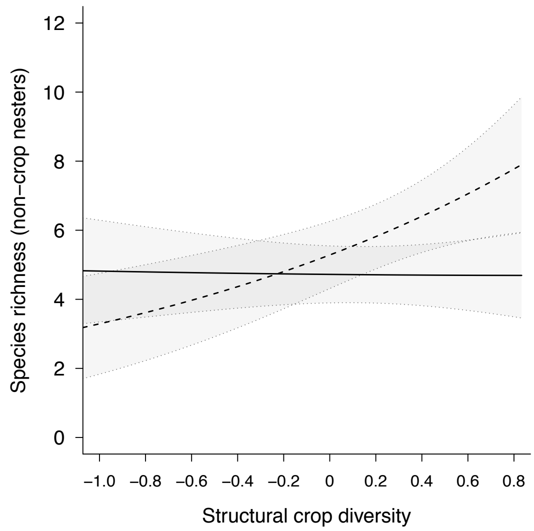


Figure 10. Estimated effect and 95 % CI of structural crop diversity (*H'*$_{structural}$) on species richness of non-crop-nesting species in relation to landscape composition. Effects were calculated by holding the proportion arable land constant at 0.4 (solid line) and 0.8 (dashed line) while varying structural crop diversity.

The spatial arrangement of crops (mean field size) was important also for farmland bird diversity (Figure 9). Species richness and, to a lesser extent, also abundance, of non-crop-nesting species were associated with smaller fields. This verifies the importance of field margin habitats for food provision for these species (Vickery et al., 2009), but possibly also reflects habitat complementation for species that require several habitats (Low et al., 2010; Siriwardena et al., 2012). In contrast, field nester abundance was higher on farms with large fields, possibly reflecting avoidance of linear features such as field edges, which attract predators (Morris & Gilroy, 2008; Schneider et al., 2012; Figure 9).

4.1.2 Non-crop habitats and landscape composition

The proportion of arable land in a 1000-meter radius related positively to species richness of field nesters, but did not affect non-crop nesters to any great extent (Figure 9). Not surprisingly, trees and shrubs had positive effects on non-crop-nesting species richness and abundance, as these structures provide important foraging and nesting substrate for this species group. However, trees and shrubs also had negative effects on field nester abundance, presumably reflecting avoidance of vertical structures in the landscape associated with increased predation risk on the cropped area of fields (Whittingham & Evans, 2004; Gabriel et al., 2010). Non-crop habitats (*i.e.*, open non-cropped areas) and in-field islands held little importance for either species group (Figure 9). Hiron et al. (2013b) also found in-field islands to support comparatively low species richness and abundance of birds, as compared to other surveyed non-crop habitats, but in contrast had higher between-site variation (beta diversity) in species richness.

4.2 Agri-environmental policy

4.2.1 CAP crop diversification

Irrespective of the biodiversity benefits of crop diversity (presented in Section 4.1.1), almost all SVFA farms already met the CAP crop diversification requirements even before they were imposed, or were exempted from them for having high proportions of rotational grass, fallow or permanent grassland (Figure 11). Evidently, the crop diversification measures in the CAP for 2015-2020 may largely be inoperative, a fact that has previously been shown to hold true across many Member States (Peer et al., 2014). Further, it can also be argued that the new regulations leave open the possibility of future reductions below levels beneficial for biodiversity, as these levels are presumably set too low to have benefits for wildlife, as judged by a panel of policy- and conservation-oriented experts (Dicks et al., 2014). Our study included few farms in this lower part of the crop diversity spectrum (Figure 11), however, which impedes strong conclusions regarding changes from one or two to three crops.

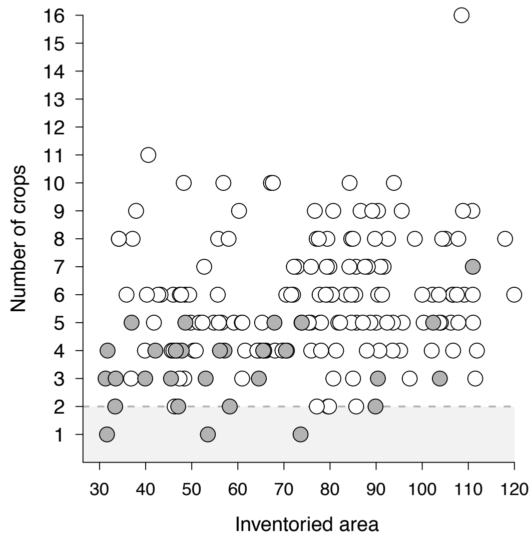


Figure 11. Crop number of the inventoried farm areas in relation to set limits in the crop diversification regulations of the CAP (three crops, shaded area). Shaded symbols: farms exempted from crop diversification for having > 75 % rotational grass, fallow land and/or permanent grassland and < 30 ha of other crops.

4.2.2 Organic farming

Results from Paper **I** corroborated the view that organic farming has relatively weak effects on farmland birds as compared to other factors, such as landscape composition and spatial arrangement of fields (Bengtsson et al., 2005; Winqvist et al., 2012; Tuck et al., 2014; Figure 9). Organic farming had positive effects only on abundance of field-nesting species and only in the most arable-dominated landscapes (Figures 9 and 12). In landscapes where arable farming was not the dominant land use, organic farms even had slightly lower field-nester densities as compared to conventional farms (Figure 12). Hiron et al. (2013a) found a very similar pattern, in the same region, where organic farming had negative effects on species richness of farmland birds in complex landscapes. This might result from high proportion of grasslands on organic farms in these landscapes, which support lower densities of field-nesting species compared to for example cereals (*e.g.*, Berg, 1993; Chamberlain & Gregory, 1999; Donald et al., 2001). Having said that, organic farming and other low-pesticide farming systems can have strong positive effects on the nesting success of birds (Boatman et al., 2004; Hallman et al., 2015), as the reduced use of pesticides increases food biomass in these systems (Girard et al., 2014; Lüscher et al., 2014). On the other hand, nesting success of some field-nesting species can also suffer from the increased field operations from mechanical weed management of organic farming (Kragten & de Snoo, 2007; Kragten et al., 2008), or from a higher risk of nest predation on organic farms, where high quantities of vertical elements such as trees attract corvids (Gabriel et al., 2010).

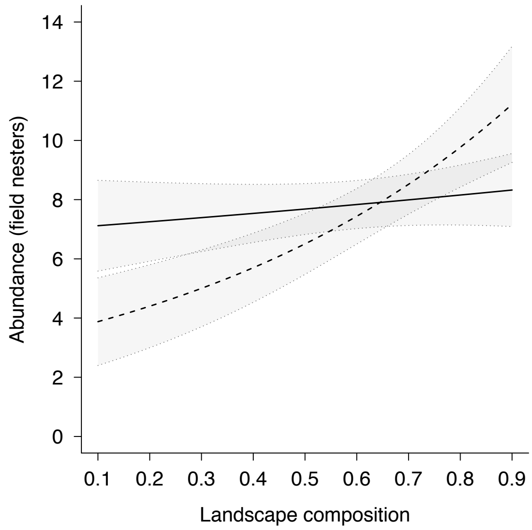


Figure 12. Estimated abundance including 95 % CI of field-nesting species on conventional (solid line) and organic farms (dashed line) along a landscape gradient (proportion of arable land).

4.2.3 Buffer strips

In Paper **II** we assessed whether fields with densely vegetated buffer strips held higher densities of territorial Skylarks and invertebrate food sources, as compared to fields without such strips. We found that fields with buffer strips supported higher (+0.51±0.26 territories/ha up to 100 meters into the field) densities of Skylarks and boosted invertebrate activity densities compared to fields without buffer strips (Figure 13). These effects were most apparent early in spring, but persisted throughout most of the sampling period (Figure 13). This suggests that buffer strips could target multiple environmental objectives on cereal fields in heterogeneous farmland, by decreasing surface run-off and also increasing biodiversity. Future research should work to identify buffer-strip management practices that further increase their value to biodiversity at the local scale, and investigate how they affect farmland biodiversity in different landscape types at larger spatial scales for more efficient implementation. For instance, managing dense swards through selective cutting of buffer strips may improve food accessibility for ground-foraging birds, while still maintaining vegetation adjacent to watercourses to sustain invertebrate populations and reduce agro-chemical runoff (Douglas et al., 2009; Vickery & Fuller, 1998).

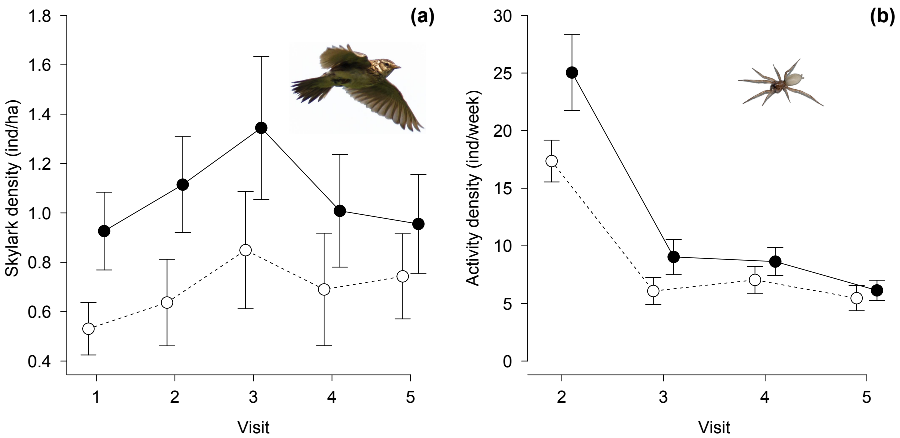
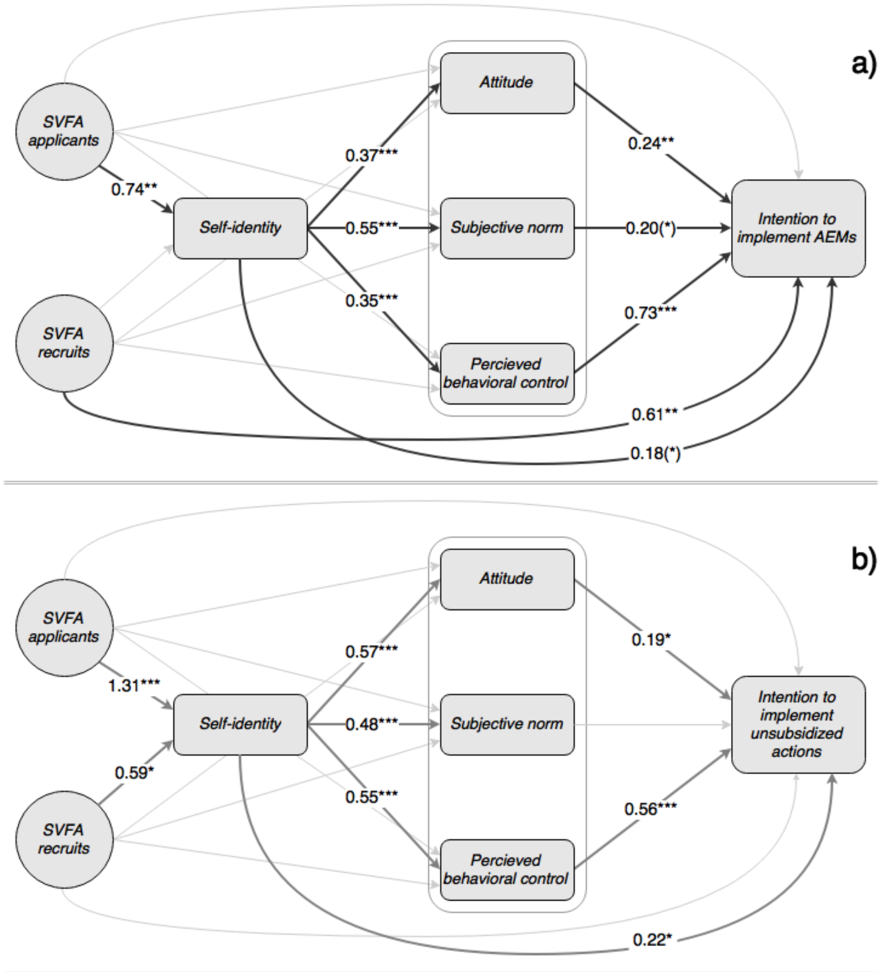


Figure 13. a) Abundance of territorial Skylarks (Mean±SE) and b) activity density of spiders in relation to presence of grass buffer strip and time in season (visit). Continuous line and closed circle, buffer strip present; dashed line and open circle, buffer strip absent.

4.3 Motivational differences between subsidized and unsubsidized conservation

Paper **III** examined if and how participation in SVFA affected farmers' intrinsic motivation to implement AEMs and unsubsidized measures. All SVFA farmers had higher intentions to implement both forms of measures compared to the control group (Figure 14). This acknowledges the suitability of collaborative approaches such as SVFA for targeting farmers that are generally unwilling to participate in AEMs (Espinosa-Goded, Barreiro-Hurlé, & Dupraz, 2013). The higher intention to implement AEMs among SVFA farmers recruited by project managers, however, was not linked to behavioral determinants (including self-identity), suggesting that SVFA did not affect farmers' attitudes and norms regarding AEMs (Figure 14). This is in line with the notion that payments, by reducing the costs of actions, inhibit development of intrinsic motivation (Herzon & Mikk, 2007). Further, Burton & Paragahawewa (2011) suggest that development of intrinsic motivation is also constrained by a lack of opportunities for farmers to display cultural capital, in the form of knowledge, learned skills and/or values, when implementing AEMs. Thus, it has been suggested that positive attitudes and norms among farmers towards AEMs might be promoted if AEMs were designed to instigate such cultural capital, for example by paying for delivery of results on set targets of conservation production, instead of area-based payments (Gibbons et al., 2011; Burton & Schwarz, 2013).

There was also a strong association between perceived behavioral control (farmers' perceived ability to perform actions) and intention to implement conservation measures (Figure 14). This stresses the importance of finding more efficient ways to transfer knowledge regarding the availability, and implementation, of conservation practices (Steyaert et al., 2007; Lauber et al., 2011; Haenn et al., 2011). Finally, this study also provided insights into the pathways by which self-identity influences intentions. For AEMs, self-identity had a strong effect on subjective norm, highlighting the importance of social factors in determining AEM engagement (Michel-Guillou & Moser, 2006). For unsubsidized conservation measures, self-identity instead had stronger effects on attitudes and perceived behavioral control (Figure 14). Possibly, this is a reflection of the importance of personal driving forces behind these measures, as they are fortified neither by subsidy policies nor by the peer group (Lokhorst et al., 2011).



(*) $p < 0.1$, * $p < 0.05$, ** $p < 0.01$, *** $p < 0.001$

Figure 14. Structural models with standardized parameter estimates, explaining the effects of the Swedish Volunteer & Farmer Alliance (SVFA) on intention to implement a) subsidized agri-environmental measures (AEMs) and b) unsubsidized conservation measures in the theory of planned behavior model augmented to include self-identity.

4.4 Impacts of the Swedish Volunteer & Farmer Alliance

4.4.1 Farmland bird diversity

In Paper **IV**, we showed that implementation of conservation measures at the farm-scale had positive effects on farmland bird species that rely on non-crop nesting habitats in the most arable-dominated landscapes (Figures 15 and 16). In these landscapes, the generally negative trend in abundance between first and second inventories (–12 % for field-nesting species and –28 % for non-crop-nesting species) was successfully moderated for non-crop nesters on farms that implemented conservation measures and on farms that implemented a sufficiently large number of measures declines were effectively stopped (Figures 15 and 16). Again, this points to the view that interventions to support farmland biodiversity have the biggest impact in the most arable-dominated landscapes (Batáry et al., 2011). In contrast, field-nesting species did not show such a response to conservation measures (Figure 15), which implies that the extent of implementation was not sufficient for these species, or that they rely on measures not promoted in SVFA. However, higher structural crop diversity increased both species richness and abundance of field nesters (see also, Henderson et al., 2009; Gottschalk et al., 2010; Eggers et al., 2011). Thus, measures for field-nesting species probably need to relate to land use in arable fields, and they should probably be implemented at larger scales to affect these species (Báldi & Batáry, 2011).

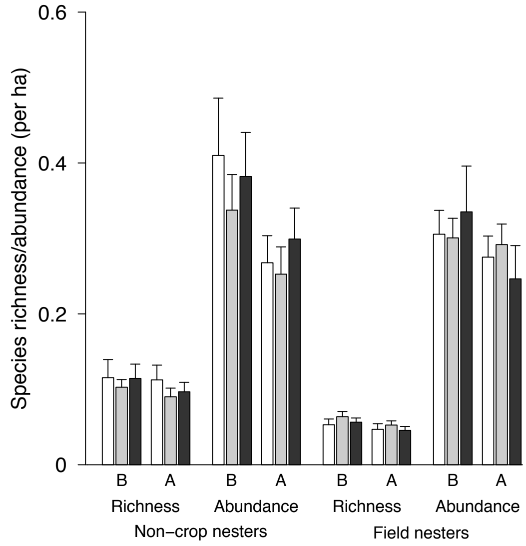


Figure 15. Species richness and abundance of non-crop and field nesters (Mean±SE) before (B) and after (A) implementation of no (white bars), 1–2 (gray bars) or ≥ 3 conservation measures (black bars).

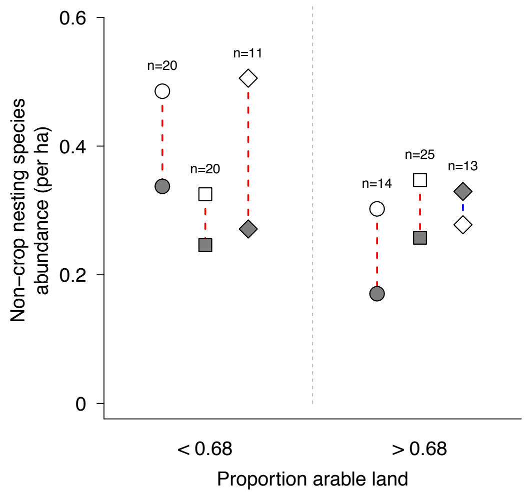


Figure 16. Abundance of non-crop-nesting species before (open symbols) and after (closed symbols) implementation of conservation measures in relation to landscape type. Circles, no conservation measures; Squares, 1–2 implemented measures; Diamonds, ≥ 3 implemented measures.

4.4.2 Implementation of conservation measures

Unfortunately, one third of the participating farms in SVFA did not implement any measures, and most farms that did implemented only few (45 % of the farms implemented between one and two measures (one measure: 25 farms, two measures: 20 farms), and 23 % implemented between 4 and 8 measures (three measures: 10 farms, ≥ four measures: 14 farms; Figure 17). This in spite of farmers' receiving a consultative visit explicitly focused on conservation advice and an increased intention to perform measures among the SVFA farms.

Thus, it is crucial to put more effort into ensuring high levels of measure implementation. Here, commitment-making strategies (Lokhorst et al., 2012), or benchmarking instruments to make farmers aware of their performance compared to neighbors (de Snoo et al., 2010), could be effective methods to increase the prospect that farmers act on inclinations to implement conservation measures.

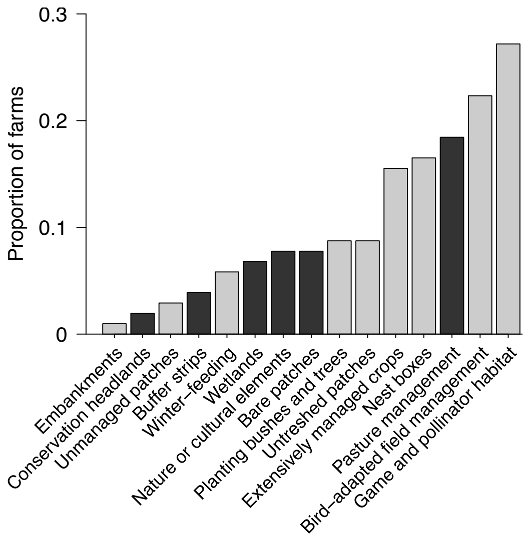


Figure 17. Proportion of farms adopting subsidized (black bars) and unsubsidized (gray bars) conservation measures implemented within the SVFA. See Table 1 for descriptions of the measures.

5 Conclusions

This thesis suggests that successful farmland biodiversity conservation partly relies on agri-environmental policies that increase the structural heterogeneity of arable landscapes (*e.g.*, through crop diversification and establishment of buffer strips). This is especially important in regions where arable farming is predominant and farmland heterogeneity is low, and similarly measures such as organic farming had positive effects only in the most arable-dominated landscapes. Further, we also found that despite potential biodiversity benefits from crop diversity of arable fields, crop diversification in the EU Common Agricultural Policy (CAP) may miss this opportunity by setting limits too low and by neglecting variability in structure and management of different crop types.

Furthermore, we confirmed the importance of landscape composition and occurrence of non-crop habitats (*e.g.*, field margins and trees and shrubs) for the farmland bird community. However, it is important to notice that effects of landscape composition (and also of crop heterogeneity) differed largely between field nesters and non-crop nesters, who have different habitat requirements.

We also showed that collaborative efforts to enhance biodiversity conservation can be useful to promote implementation of agri-environmental measures (AEMs) and unsubsidized conservation measures. In the Swedish Volunteer & Farmer Alliance, these implemented measures had positive effects on farmland bird diversity, but these effects were limited by low implementation rates and restricted to the most intensively farmed landscapes. We also showed that farmers preferred to perform unsubsidized conservation measures, which contributed to positive effects on bird diversity. However, despite these limitations, SVFA is a noteworthy model of collaborative biodiversity conservation, not least for its magnitude in terms of the number of farms involved and area covered.

Our results suggest that collaborative biodiversity projects, in farmland and elsewhere, could potentially benefit from considering methods to further improve rates of implementation and instigate long-term behavior change (such as commitment-making and/or benchmarking). Also, covering larger portions of intensive arable regions, collaborative approaches may still prove a viable alternative to more centralized solutions of farmland biodiversity conservation in such landscapes. Thus, we suggest that volunteer-based approaches are a viable alternative to centralized solutions of farmland biodiversity conservation, but that these projects could benefit from a better recognition of the importance, and costs, of managing and following up stakeholders' involvement as well as monitoring environmental outcomes.

We also found that underlying motivations behind AEMs and unsubsidized conservation differed, where social factors was most important for farmers' engagement in AEMs, while unsubsidized conservation was driven more by personal valuations and interests. Consequently, current AEMs that impede production may have poor chances of becoming accepted by farmers in farming-intensive regions with prevailing productivist norms. We conclude the future of AEMs for biodiversity protection partly lies in better integration into the cultures of farming communities, alongside an evidence-based approach to biodiversity conservation receiving the same political bearing as, for example, issues of trade liberalization or subsidies without production restrictions (such as biofuel subsidies).

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