MASTER THESIS

Course code: BIO5010 Name: Jonas Hovd

Distribution and Habitat Preferences of Eight Common Farmland Bird Species in Central Norway

Date: 29.05.20 Total number of pages: 41

NORD University

www.nord.no

Distribution and Habitat Preferences of Eight Common Farmland Bird Species in Central Norway

Jonas Hovd

Nord University

Faculty of Biosciences and Aquaculture

2020

Summary

Many farmland bird populations have declined severely over the past 60 years. Throughout western Europe most of the declines are considered to be the result of agricultural intensification, farmland abandonment and landscape homogeneity. Farmland consists of many habitat types and birds can be grouped based on their use of these biotopes. A bird is considered a farmland bird when more than 50% of its population nest in agricultural habitats. Monitoring indicator species has been a cost effective and well-utilized method for measuring ecosystem health. While farmland bird distribution and habitat preference are well-researched topics in western Europe, there are only a few research papers published regarding farmland bird populations in northern Europe.

This study examined the habitat preferences and distribution of eight common farmland bird species in Norway. The analyses were based on presence/absence data of each of the eight species along with location data collected from field surveys in the years 2013, 2014 and 2019, covering 538 survey plots spread evenly across nine study locations in central Norway. Data was analysed using computer software such as GIS and R to identify abundance and richness in the nine study locations, as well as the proportion of landcover, landscape complexity and landscape diversity of the same locations. Logistic regression models were run for each of the eight farmland bird species, with the presence of the species as response variables and habitats as explanatory variables.

The analysis showed that locations in the midlands had a higher presence of farmland bird species than at the coast and mountains. Analysis of landscape cover showed that the locations in the midlands had the greatest proportions of fully cultivated soil, while the coast was more dominated by open land and the mountains more dominated by forest. The abundance and richness of farmland bird species were highest in the midlands and in the southern parts of the coast. Landscape complexity had overall low mean values in the midlands, high values at the coast and values ranging from high to low in the mountains. The analysis of landscape diversity showed higher diversity at the coast, lower diversity in the mountains and more varying landscape diversity values in the midlands. The logistic regression models of the eight farmland bird species showed that most of the final models had low model fit, except for two species. However, most of the final models showed a positive effect of agricultural habitats, mainly fully cultivated soil and pastures. Most of the eight farmland bird species in the study proved to be good indicator species of farmland habitats in central Norway. Additionally, this study showed the importance of suitable habitats, landscape complexity and landscape diversity for farmland birds in the agricultural landscape.

Preface

I want to give a special thanks to my main supervisor Jan Eivind Østnes, Associate Professor of the faculty of biosciences and aquaculture at Nord University, for inviting me to the project and for being a great guide and mentor throughout the whole process. I also want to give a special thanks to my second supervisor, Amy Elizabeth Eycott, Associate Professor of the faculty of biosciences and aquaculture at Nord University, for her priceless statistical competence and guidance, and for being a selfless provider of feedback, motivation, and laughter. For their contributions to my master's thesis I would also like to thank Magne Husby, Torgrim Sund and Sam Steyaert.

Table of Contents

1. Introduction

European bird populations, particularly farmland birds, have declined severely in distribution, abundance and biomass, over the past 60 years (Donald *et al.*, 2001; Inger *et al.*, 2015). Several factors have negatively influenced the farmland bird populations, acting across borders and different agricultural practices. Throughout Western Europe most of the declines are considered to be a result of intensification of agricultural production (Donald *et al.*, 2001; Donald *et al.*, 2006), but also abandonment of farmland (Wretenberg *et al.*, 2007) and increased landscape homogeneity (Heikkinen *et al.*, 2004; Pickett & Siriwardena, 2011; Wretenberg *et al.*, 2010). The effects of climate change may also exacerbate these trends. Environmental changes over large spatial scales due to climate change seem to drive population change, and may increase the risk of extinction (Post & Forchhammer, 2004; Root *et al.*, 2003; Walther *et al.*, 2002). Due to these negative prospects the need to identify the main factors that cause population decline in birds increases (Bennett *et al.*, 2014). Further, these declines could impact other ecosystem functions and consequently impair the delivery of important services (Şekercioğlu *et al.*, 2004). For example, birds provide pest and parasite control, seed dispersal and aesthetic value (Anderson *et al.*, 2011; Geiger *et al.*, 2010; Wenny *et al.*, 2011; Whelan *et al.*, 2008; Winqvist *et al.*, 2011). The population collapses that are witnessed can lead to population extinctions and further collapse of important ecosystem services. Searching for and being aware of these negative drivers to bird population are therefore crucial (Pedersen & Krøgli, 2017).

Farmland consists of different habitats, and farmland birds can be grouped based on their use of these biotopes in the agricultural landscape (Solheim, 1993; Thingstad & Vie, 1995). A bird is regarded as a farmland bird, when more than 50 % of the population breeds in cultural landscape (Pedersen & Krøgli, 2017). Cultural landscape is defined by Engan *et al.* (2008) as the total area which is influenced by recent or former agricultural activity and settlement. The cultural landscape consists of areas which are fully or surface cultivated and pasture (Engan *et al.*, 2008).

Factors that seem to have been key drivers for change in distribution and abundance of farmland birds are land use and land use change (Siriwardena *et al.*, 1998). Also, recent studies in European landscapes have shown that response to land use intensification or abandonment are depended on the landscape context. For example, Fox (2004) found stable or increasing populations of farmland birds in spite of agricultural intensification. Other studies have shown that in areas where the amount of arable land is large, farmland birds seemed to benefit from extensively managed areas, whilst in areas where the amount arable land were small, several farmland birds species responded positively to agricultural intensification (Robinson *et al.*, 2001; Wretenberg *et al.*, 2010). The declining population trends could also be explained by reduced farmland landscape heterogeneity. Positive effects have been found from more diverse landscape types implemented in the farmland. For example set-aside areas (Gillings *et al.*, 2010; Henderson *et al.*, 2012), hedgerows and field margins (Batáry *et al.*, 2010; Bradbury *et al.*, 2000) and, in some cases land abandonment (Woodhouse *et al.*, 2005). This is because many farmland birds require several different habitat types to fulfil their niche requirement throughout the breeding season (Dallimer *et al.*, 2012).

Norway has joined several international treaties and conventions that aims to secure biological diversity (Engan *et al.*, 2008). Farmland birds in Norway has had a similar population decline as in western Europe (Dale & Hardeng, 2016; Husby & Reinsborg, 2015; Kålås *et al.*, 2014; Shimmings & Øien, 2015), and many species have been red-listed (Henriksen & Hilmo, 2015). Engan *et al.* (2008) states that the drivers for change in biological diversity in the Norwegian agricultural landscape are abandonment, intensification of farmland production and nature reclaiming pastures and meadows. Pedersen and Krøgli (2017) claims that in agricultural landscapes with spatial variation of farmland, meadows, trees and shrubs, and natural vegetation, open water and forests, had a positive effect on the diversity of farmland bird species, but a negative effect on the total number of individuals. They argue that a higher landscape diversity might provide more habitats and more niches to sustain a higher amount of farmland bird species, but as the areas of suitable habitat becomes smaller, the number of individuals of each species it is able to sustain lowers.

Farmlands in Norway are, compared to farmlands in other countries in Western Europe, relatively extensively managed with a high level of landscape heterogeneity (Pedersen & Krøgli, 2017). However, many of the farmland birds breeding in Norway have their wintering areas in continental Europe on shared winter ranges with birds migrating from other countries. Birds that share common wintering grounds could experience similar trends in populations across countries (Wretenberg *et al.*, 2006). Migrating birds that travel between countries or continents could cause complications in identifying drivers of population change, especially where the condition of the breeding grounds differs markedly between each country or continent. For example, two populations of the same species migrate to the same wintering grounds, but one population breeds in an extensively managed farmland and another population breeds in an intensively managed farmland. There is a knowledge gap for populations that migrate between such countries, and there are few studies that have been published on farmland birds and land use in Norway (Pedersen & Krøgli, 2017). Wretenberg *et al.* (2006) argues that there is a need to study populations outside continental Europe, particularly in countries that are dominated by extensively agricultural practices and where the birds migrate to a common wintering ground.

Biological diversity is defined as the sum of species diversity, genetical diversity and ecological diversity, and is practically not possible to measure in perfect detail (Hilty & Merenlender, 2000; Landres *et al.*, 1988). Instead, indicators can be used to get an insight into the biological diversity. These indicators can be used to observe changes in nature and its state. An ideal indicator would, for example, be a group or a single species where the total population trend would be representative of the mean population change for all the species in an ecosystem, also species from other taxa, and thus could work as a measurement of the ecosystem's health (Gregory & Van Strien, 2010; Gregory *et al.*, 2005). Indicators like these do not usually exist, but a chosen set of species used as indicators could provide a useful insight into the changes in nature (Kålås *et al.*, 2014).

Birds have many traits that make them ideally as environmental indicators (Gregory & Van Strien, 2010), and changes in bird populations are regarded as a good indicator on the state of ecosystems (Kålås *et al.*, 2014). These traits are for example that 1) birds responds to threats, 2) birds represent different levels in the food chain, 3) birds are found in all nature types, 4) there is an already well-developed knowledge of species, ecology and methods for population counts, 5) there is a large network of people with interest and knowledge which can perform counts, and 6) birds are used as indicators for ecologic sustainability in the EU and in many European countries (Kålås *et al.*, 2014).

When deciding indicator species for various types of habitats, it is important to include species which are representative for these nature types, and that there exist data of good quality to evaluate population changes over a time period. Husby and Kålås (2011) selected eight bird species as indicators for farmland based on the following criteria: i) that a large part of the population is found in farmland during the breeding season (>80 %), and ii) that there is adequate quantitative population data from the surveillance project TOV-E. The species included were northern lapwing *Vanellus vanellus*, eurasian curlew *Numenius arquata*, eurasian skylark *Alauda arvensis*, barn swallow *Hirundo rustica*, whinchat *Saxicola rubetra*, common starling *Sturnus vulgaris*, yellowhammer *Emberiza citronella* and white wagtail *Motacilla alba.*

The common farmland birds indicators in Norway, such as northern lapwing, eurasian curlew, eurasian skylark, barn swallow, whinchat, common starling and yellowhammer has through the last few decades had an severe decline in population numbers (Gjershaug, 1994; Kålås & Byrkjedal, 1981; Kålås *et al.*, 2014; Shimmings & Øien, 2015). For example, the population of northern lapwing was estimated to be 40 000-80 000 pairs between the years 1970-1990 (Gjershaug, 1994). More recent counts has estimated the population to have declined to 7 500-10 000 pairs (Heggøy & Øien, 2014; Shimmings & Øien, 2015).

Although the population of barn swallow declined from 1970 to the 1990s (BirdLife, 2004; Gjershaug, 1994), recent estimates has shown a slight increase in population in Norway and Sweden (Green *et al.*, 2016; Kålås *et al.*, 2014; Shimmings & Øien, 2015). White wagtail is a farmland bird indicator which not has shown clear population change during the last decades (Shimmings & Øien, 2015).

A Norwegian national representative surveillance of terrestrial breeding species was established in 2005 after two years of testing in central Norway (Kålås & Husby, 2002, 2011). The project is included in “program for terrestrial nature surveillance (TOV)” which is run by the Norwegian Environment Agency and the Norwegian institute for nature research (NINA) and is called “Extensive monitoring of breeding birds” (TOV-E) (Kålås *et al.*, 2014). One of the main reasons for establishing such program was to have a nation-wide and representative measurement for changes in bird populations in the Norwegian nature, and as indicator for sustainability in Norway.

In the period 1996-2013, Kålås *et al.* (2014) found that of 55 selected terrestrial bird species, there were a significant population decline for 19 species and a significant population increase for four species. The nine species with the most severe decline include three common farmland species, viz eurasian skylark, eurasian curlew and northern lapwing. The species with a significant increase in populations were short distance migrating birds from central Europe such as common whitethroat *Sylvia communis* and eurasian blackcap *Sylvia atricapilla.* They also observed that many of the species with significant population change have shared traits. Of the 25 species in decline four species (northern lapwing, eurasian curlew, eurasian skylark and yellowhammer) are commonly found in farmland and eight species were commonly found in relation to mountains or in mountainous landscape. Shimmings and Øien (2015) published a rapport describing the population estimates for breeding birds in Norway. Out of 255 species, 57 bird species (22 %) are in decline, and some of these species have a severe decline. Especially for some waders and sea birds the situation is worrying.

The purpose of this study is to investigate the distribution and habitat preference of eight common farmland bird species in central Norway, in order to further the understanding of farmland bird populations in relation to their habitats in an area outside continental Europe. The eight bird species included in the study are those selected by Husby and Kålås (2011) as indicator species for farmlands, i.e. northern lapwing, eurasian curlew, eurasian skylark, barn swallow, whinchat, common starling, yellowhammer and white wagtail. The research questions are as following: 1) Do species presence, richness, and abundance, and landcover, habitat- complexity and diversity vary between the nine study locations? 2) Is the presence of the selected indicator species more strongly affected by a habitat type than another?

2. Materials and methods

2. 1. Study area and study design

The study was performed in Trøndelag county in central Norway. Trøndelag has a varied climate and ranges from southern boreal vegetation in the southwestern parts (i.e. Bjugn/Rissa/Leksvik) to more alpine vegetation zones in the north-eastern parts (i.e. Lierne) of the county, with middle boreal and northern boreal zones ranging between these zones (Moen, 1998). The study design was inspired by the TOV-E project that aims to survey Norwegian bird populations (Kålås & Husby, 2011).

Farmland birds were counted on a total of 540 survey plots with a unique identification number. The survey plots were selected in ArcGIS by placing a quadratic grid throughout the whole county with a distance of 1 km between each survey plot (figure 1). Nine study locations were then selected in a 3 x 3 arrangement, i.e. three rows going from south to north, and three rows going from west to east (figure 2). For each location 60 survey plots and around 10 reserve plots were randomly selected using ArcGIS. The criteria set for selection of the of the survey and reserve plots were 1) the plot was located on farmland and 2) a maximum distance of 200 m from the plot to the nearest road (figure 1). Reserve plots were only included in the study if any of the 60 survey plots in each of the nine different study locations were inaccessible.

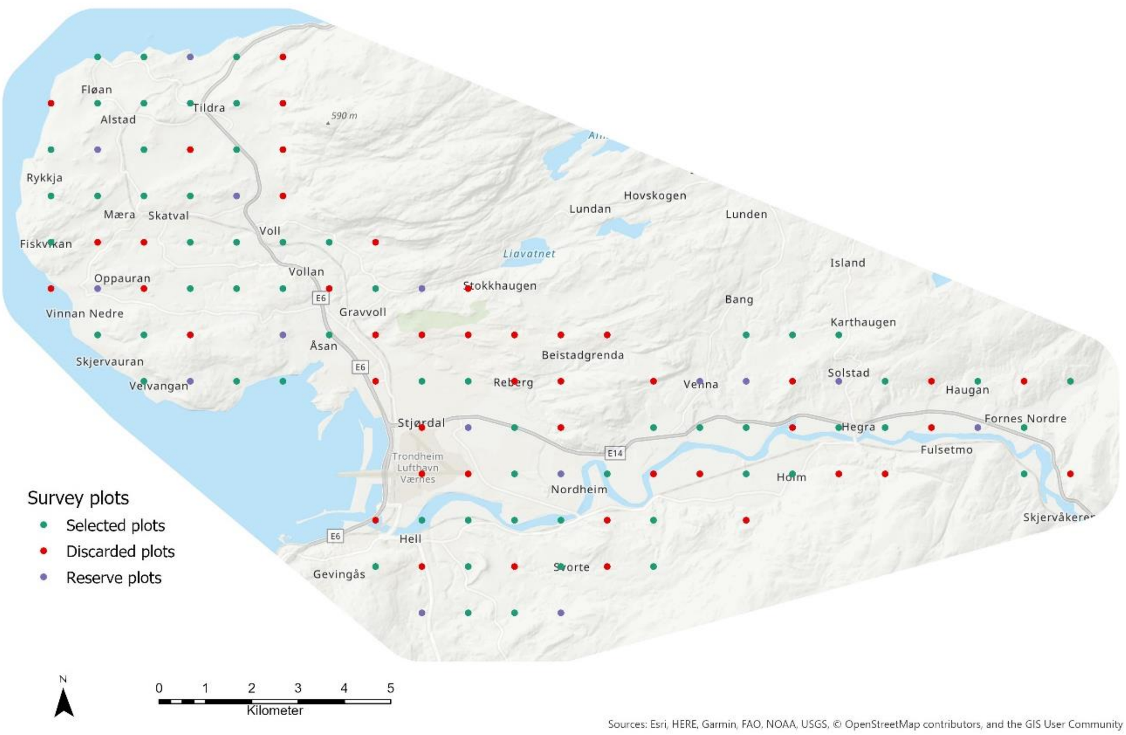

Figure 1. Example of grid and selection of discarded plots, unused reserve plots and 60 survey plots in the study location Stjørdal (southern midland area).

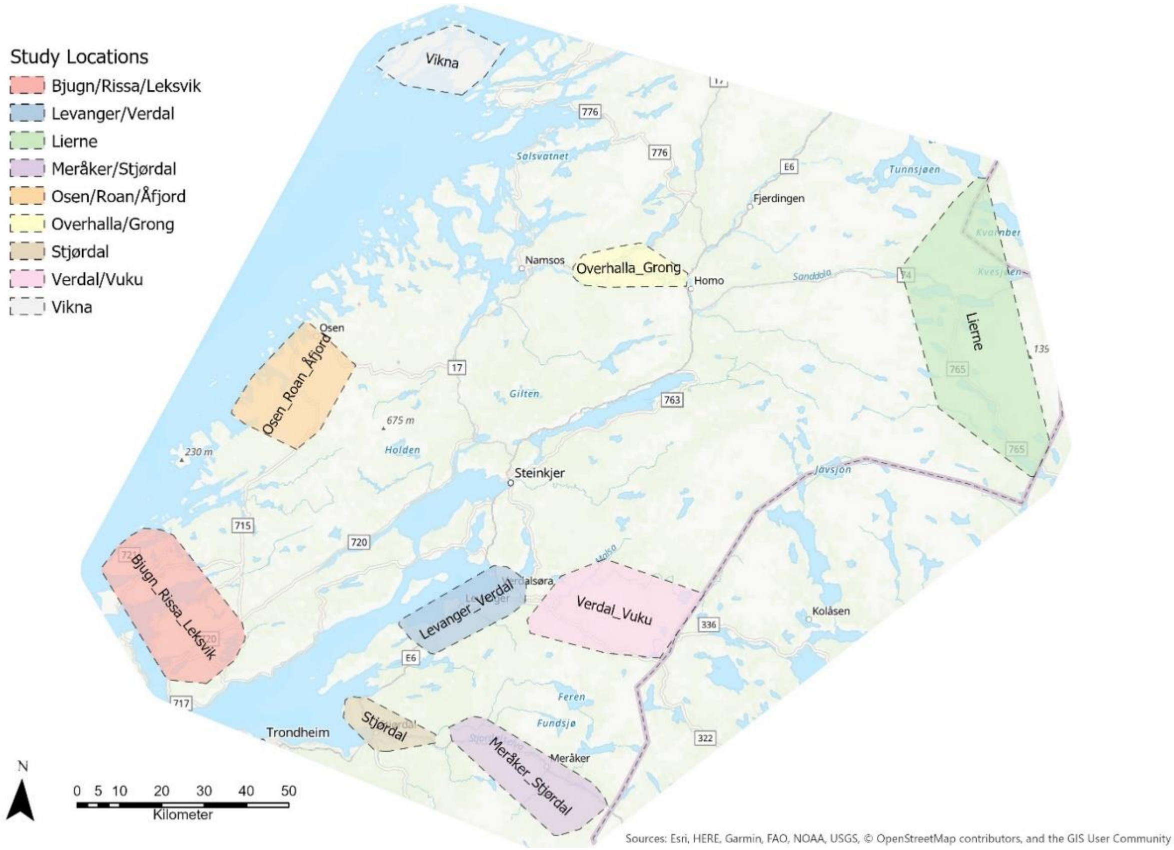

Figure 2. The nine locations in the study area positioned in a grid of 3 x 3, from south to north and from coast to mountains. The study area is in Trøndelag county in central Norway.

2. 2. Field methods

The eight selected farmland birds were counted on the 540 survey plots in the years 2013, 2014 and 2019, by using a plot sampling method. Fieldwork was performed between 15th May to 13th June. The counting plots were located by participants using maps and GPS-coordinates. Fieldwork was performed between the hours 04:00 and 11:00.

Participants of the fieldwork aimed to make the counts as close to the survey plots as possible even if the exact position could not be reached, i.e. they were not allowed to enter the farmlands. The time used on each counting plot was exactly five minutes. Within this time the participants recorded the numbers of each of the eight selected farmland bird species observed by either sight or song. Observation of a specific species, e.g. the territorial song of a male, was counted as a pair of that species. In addition to bird registration the participants also noted the type of farmland (cereal, grass, pasture, cereal/grass, other) and evaluated the condition (good/bad) of the area around the plot.

Two of the 540 survey plots were excluded from further analyses since they were not accessible in all the three years of survey, leaving a total of 538 survey plots. The two plots that were excluded were both located in the location Verdal/Vuku (figure 2).

2. 3. Variables included in GIS-analyses

The landscape data was NIBIO's FKB-AR5 dataset, analysed in ArcGIS Pro (ESRI, 2019). A buffer analysis was performed using the 538 survey plots as locations for the analysis and the FKB-AR5 dataset as the landscape area data. The radius of the buffer analysis was set to 500 m so it would not overlap with adjacent buffers, as the survey plots were set one kilometer apart. These 500 m buffers were used to extract area data from the FKB-AR5 dataset from their corresponding location. The data from the buffer analysis was then extracted to Excel where the area data were pivoted and connected to the bird counting data as area variables for each counting plot.

The habitat variables used in the analysis were classified and described by AR5 classification system, provided by NIBIO (Ahlstrøm *et al.*, 2019). The landscape type is determined by the areas actual state. These landscape types as variables are presented in table 1.

Table 1. Landscape variables with definitions given by Ahlstrøm *et al.* (2019).

Variable	Definition
Fully cultivated soil	Areas which are cultivated to normal plough depth, and can be used for *poaceae* (grass types) or as meadow, and is renewable by ploughing
Surface cultivated soil	Agricultural landscape, which is mostly processed on the surface and able to be mechanically harvested.
Pasture	Agricultural areas which could be used as grazing land, and not able to be processed with machines. Where 50 % of the area must be covered by agricultural grass types and graze resistant herbs.
Forest	Area with at least 6 trees per acre which is or can reach a height of 5 meter, and that is evenly spread in the area.
Marsh	Area consisting of marsh vegetation with a peat layer of minimum 30 cm.
Open land	Area, which is not a marsh, and not agricultural area, forest, built-up area, or infrastructure.
Fresh water	Consists of lakes, rivers, and streams.
Sea	Sea and ocean.
Infrastructure	Area used for infrastructure (etc. roads, railways.)
Built-up areas	Areas which are developed, or highly altered (e.g. housing, industrial area.), also adjoining areas in relation to the built-up area.
Not specified	Areas which are registered as uncertain areas.

The farmland bird species in the study consisted of eight species and data of these species were included as variables. The data of these variables include counts of pairs from the years 2013, 2014 and 2019, and registrations of presence/absence data for each survey plot from all the years combined. The counts of pairs are numerical variables while the presence/absence data are binary categorical variables.

The variables for species abundance and richness were made from the species data. Abundance is the relative representation of the eight different farmland bird species observed on the counting plots in the years 2013, 2014 and 2019 (Preston, 1948). In this study abundance is measured as the number of pairs found per counting plot. Species richness is defined by Colwell (2009) as "the number of species in a community, in a landscape or seascape, or in a region". In this study, species richness is a variable that describe how many of the eight farmland bird species that were present on the survey plots at least once during the three years of surveys.

Additional variables were made from landscape data obtained by GIS analysis in combination with RStudio. These variables consisted of the coast to mountain gradient, the south to north gradient, the fractal dimension index and the Shannon diversity index. The two geographical gradients are categorical variables describing the location of the survey plots in the study area (table 2).

Table 2. The nine study locations in relation to the coast to mountain gradient and the south to north gradient.

Gradients	***Coast***	***Midland***	***Mountain***
North	Vikna	Overhalla	Lierne
Central	Osen/Roan/Åfjord	Levanger/Verdal	Verdal/Vuku
South	Bjugn/Rissa/Leksvik	Stjørdal	Meråker/Stjørdal

The fractal dimension index used in the study is from O'Neill *et al.* (1988) and is an index of the complexity of shapes in the landscape. This index is a measure of the fractal geometry of the landscape (Mandelbrot, 1983). The index is estimated by performing a regression of the polygon area to the perimeter of each patch on a digitalized map. The fractal dimension is linked to the slope of the regression S, by relationship (Lovejoy, 1982):

$$FRAC = 2\,S.$$

If the composition of the landscape consists of simple geometric shapes such as squares, the index values would be small. If the landscape consisted of many areas with complex shapes, the index value would be large (Krummel *et al.*, 1987). In this study, the fractal dimension index values are made of spatial vector data from the buffer analysis managed in ArcGIS Pro (ESRI, 2019).

Shannon diversity index is a common index used to characterise species diversity in a community (Magurran, 1988), and accounts for abundance and evenness of the species present. The Shannon index is calculated as follows (Shannon, 1948):

$$H' = -\sum_{i=1}^{R} p_i \ln p_i$$

Where i is the amount of different landscape types in an area and p_i is the proportion of each landscape type (Shannon, 1948). The Shannon index was used as a variable describing landscape diversity in the buffer zones of the survey plots and will be designated as landscape diversity in this study.

2. 4. Statistical analysis

All statistical analyses were performed using the statistical software R (R Core Team, 2019) with necessary packages as described in the following paragraphs.

2. 4. 1. Two-way ANOVA test of richness, abundance, fractal dimension index and Shannon Index

The parameters for species richness, species abundance, fractal dimension index and Shannon index were tested separately as response variables in a two-way ANOVA test (Gelman & Hill, 2006) with the coast to mountain gradient and south to north gradient as explanatory variables. The aim was to test for differences between the response variables and the different exploratory variables. The residuals of the data were checked for normal distribution and homoscedasticity. Some of the two-way ANOVA tests had to be followed by a Tukey's post hoc comparison of least-square means, as they showed a significant differences in the main effects and in the interactions (Tukey, 1949).

2. 4. 2. Multiple logistic regression for each of the eight farmland bird species

Several variables were included to model habitat attributes for the farmland bird species. These were species presence data from 2013, 2014 and 2019 combined, landscape area variables (table 1) and statistically made variables (Fractal dimension- and Shannon index of landscape).

Each of the eight farmland bird species were analysed in a bi directional stepwise logistic regression analysis (Queen *et al.*, 2002). The presence/absence data for each species was run as a response variable, whilst the landscape area variables (table 1) and the variables for landscape complexity and landscape diversity were included as explanatory variables. The landscape variable "not specified" (table 1) was excluded from the models since the area values of this variable was small and only present in two plots out of the 538 survey plots. The step procedure tested all the different model compositions and presented the models with the lowest AIC score for each of the eight farmland bird species tested. AIC is defined as

$$AIC = -2(\text{log-likelihood}) + 2K$$

Where K is the number of model parameters (the number of variables in the model plus the intercept) and log-likelihood is a measure of model-fit, the greater the value the better the fit (Akaike, 1974; Burnham *et al.*, 2011).

The significance of the variables included in the final regression model was determined by the Wald test. The Wald chi-squared test is used to test if the response variables in a model are significant, i.e. in this case significant means that the variable adds something to the model, while variables that add nothing can be deleted and would not affect the model in a meaningful way (Agresti, 2003). The Wald test statistic formula is

$$W_T = \frac{[\hat{\theta} - \theta_0]}{1/I_n(\hat{\theta})} = I_n(\hat{\theta})[\hat{\theta} - \theta_0]^2$$

Where $\hat{\theta}$ is the Maksimum Likelihood Estimator (MLE) and $I_n(\hat{\theta})$ is the expected Fisher information evaluated in the MLE (Agresti, 2003).

To evaluate the models beyond the AIC score it could be necessary to perform a test of model evaluation such as the likelihood ratio test against null model equivalents of the models. Likelihood ratio test compares the goodness of fit of two models and is defined as such (MacKenzie *et al.*, 2017)

$$LRT = -2\ln\left(\frac{L(\hat{\theta}_0|x)}{L(\hat{\theta}_A|x)}\right)$$

Where $\hat{\theta}_0$is the simpler model and $\hat{\theta}_A$ is the model with more parameters (MacKenzie *et al.*, 2017).

A McFadden pseudo-R^2 (*ρ^2)* test was also performed. This test tests the model fit by measuring how well the variables of the model explain an outcome, i.e. if the test value is low, there are other unexplained variables that could explain the outcome in a better way (Long & Freese, 2006). McFadden's ρ^2 measure is defined as

$$R^2_{McFadden} = 1 - \frac{\log(L_c)}{\log(L_{null})}$$

Where L_c is the likelihood value from current fitted model, and L_{null} is the corresponding value for the null model (model with only an intercept and no covariates)(Hosmer Jr *et al.*, 2013).

3. Results

During the three years of survey at least one species of farmland birds was recorded on 484 of the 538 surveyed plots. The mean number of species on each plot varied from 1.15 in 2019 to 1.53 in 2014. For all three years combined there was a mean species richness of 2.48. The total observations of farmland bird pairs varied from 818 in 2019 to 1082 in 2014.

3. 1. Species presence of all the years per location

Species presence on the total number of survey plots per study location differed markedly between the locations (figure 3). The locations in the mountainous part of Trøndelag generally showed a lower presence of farmland bird species per survey plot than the locations in the midlands and coast. Vikna had the lowest presence of species with two of the eight species completely absent, i.e. northern lapwing and eurasian skylark. Northern lapwing was also almost absent in the other coast locations and, in addition to Vikna, it was also absent in the location Osen/Roan/Åfjord. The eurasian skylark was almost absent in the mountain locations with a presence only in one of the three locations (Lierne) and had also a low presence in the coastal locations. Two of the midland locations (Stjørdal and Levanger/Verdal) had the highest presence of farmland birds. In these two locations the yellowhammer was observed on almost 100 % of the survey plots.

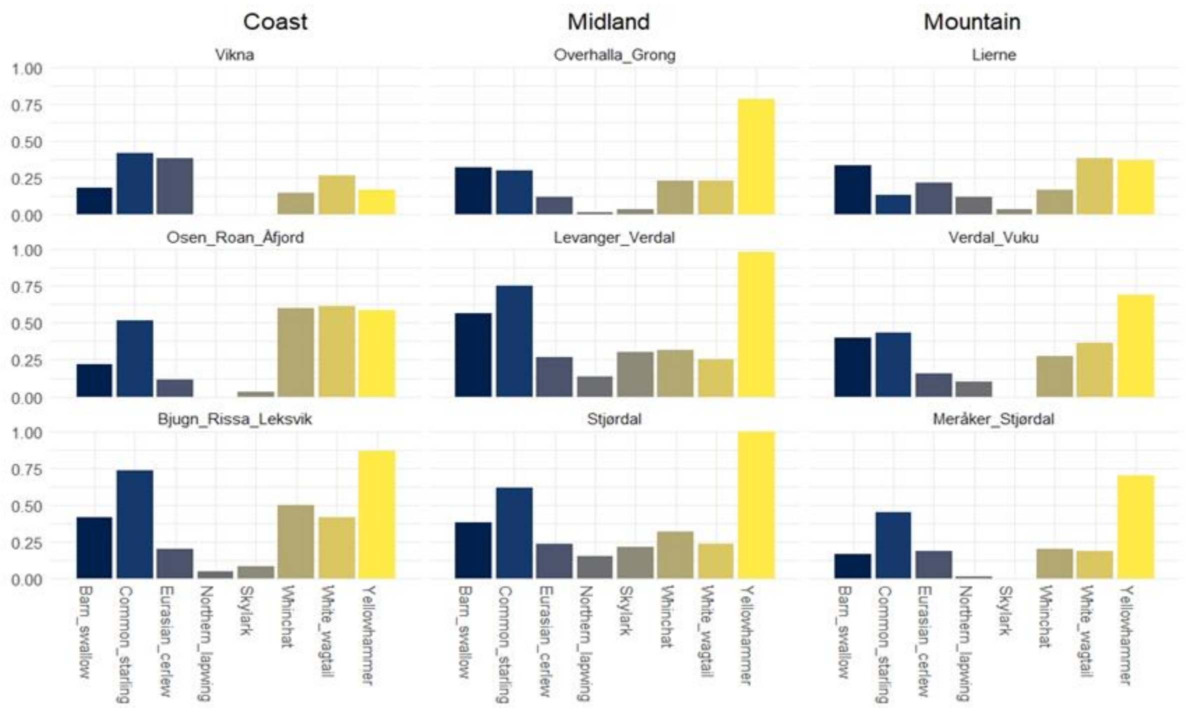

Figure 3. Total presence of eight farmland bird species on the survey plots during the years 2013, 2014 and 2019. The survey plots are grouped in nine different locations and are aligned in the figure corresponding to their placement on the coast to mountain gradient, and their placement on the south to north gradient, with the bottom row being south, and top row being north.

3. 2. Land cover and distribution

The three dominant landcover types in the study were forest, fully cultivated soil, and open land (table 3), and these covered 79 % of the total area of the buffer zones of the survey plots. The other landcover types each constituted less than 5 % of the total area.

Table 3. Total landcover area (m^2) and percentage of total cover of the survey plots buffer zones.

Landscape type	Sum area (m^2)	Percent of total m^2
Built up area	15093	3.6 %
Forest	158029	37.5 %
Fresh water	19357	4.6 %
Fully cultivated soil	125758	29.9 %
Infrastructure	6145	1.5 %
Marsh	18217	4.3 %
Not specified	284	0.1 %
Open land	47748	11.3 %
Pasture	12225	2.9 %
Sea	17325	4.1 %
Surface cultivated soil	949	0.2 %

Landcover and distribution differed both along the coast to mountain gradient and the south to north gradient (figure 4). The coastal locations differed from the midland and mountain locations and were to a larger degree dominated by open land. The proportion of open land also increased towards north. Midland locations had a higher proportion of fully cultivated soil, and the largest overall cover of fully cultivated soil was in the central midland. The mountain locations had a higher proportion of forest cover than the midland and coastal locations. The proportion of fully cultivated soil decreased from south to north.

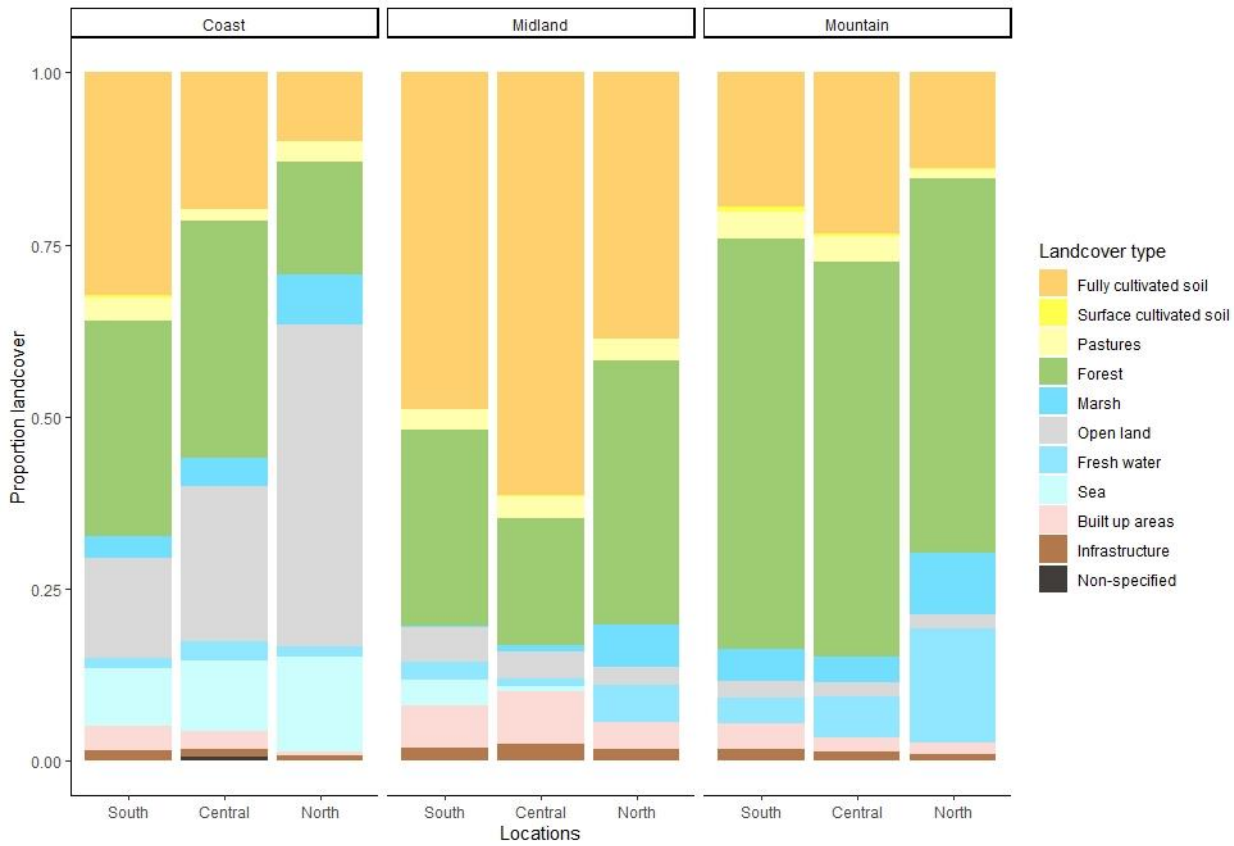

Figure 4. Proportion landcover type distribution for the nine study locations of the coast to mountain gradient and south to north gradient. The proportion of fill in the graphs indicate the total proportion of landcover types in the study locations.

3. 3. Species richness in relation to location

Species richness might be dependent on the landscape types within a habitat, but may also differ by location on a coast to mountain gradient and a south to north gradient. A two-way ANOVA was used to test the farmland bird species richness on gradients ranging from coast to mountains (coast gradient) and south to north (north gradient). Species richness was significantly different over the coast to mountain gradient ($F_{2,529}$= 19.35, p < 0.001, η^2 = .058), with an overall lower species richness in the mountains (figure 5A). Species richness was also consistently lower in the northern locations ($F_{2,529}$= 35.84, p < 0.001, η^2 = .108) (figure 5A). The interaction between the coast to mountain gradient and south to north gradient was also significant ($F_{4,529}$= 6.30, p < 0.001, η^2 = .038): The northern part of the coast had the lowest mean farmland bird species richness, whereas the location with the highest mean species richness was the central midland (figure 5A).

The main effects and the interaction term of the test were significantly different. The post-hoc test of least square means of species richness (table 4) showed that the central midland (lsmean = 3.57) was the group that was significantly most different from north coast (lsmean = 1.57). The groups with the highest least square means (i.e. the most different from the overall mean) were all located in the central and south of the midland and coastal locations.

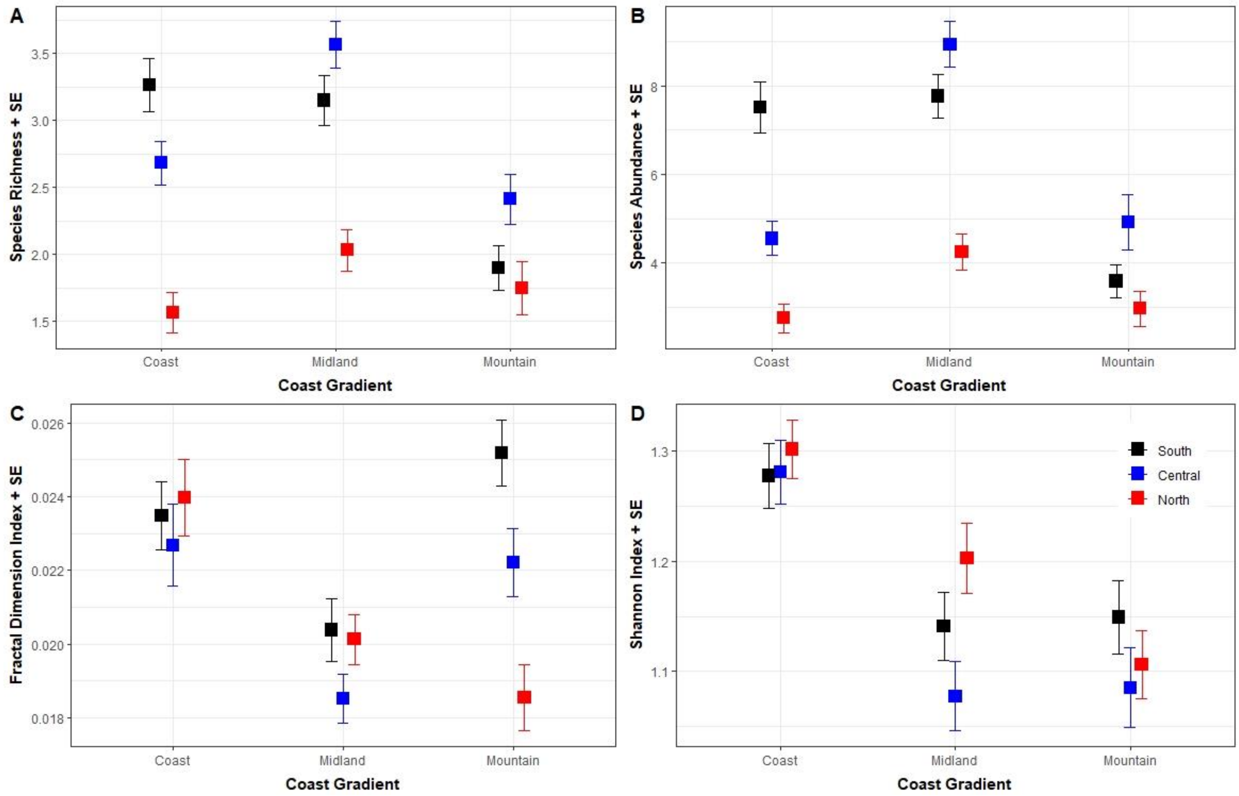

Figure 5. Interaction plots of the means and standard errors of A) Species richness, B) Species abundance, C) Fractal dimension Index (Landscape complexity) and D) Shannon index (Landscape diversity) on the coast gradient and north gradient.

Table 4. Post-hoc test of species richness for the nine study locations, with least square means, standard errors, degrees of freedom, upper and lower confidence interval, and least square mean groups. Means sharing a letter are not significantly different.

North Gradient	Coast gradient	Lsmean	SE	Df	Lower.CL	Upper.CL	Group
North	Coast	1.57	0.176	529	1.08	2.06	A
North	Mountain	1.75	0.176	529	1.26	2.24	A B
South	Mountain	1.90	0.176	529	1.41	2.39	A B
North	Midland	2.03	0.176	529	1.54	2.52	A B C
Central	Mountain	2.41	0.179	529	1.92	2.91	B C D
Central	Coast	2.68	0.176	529	2.19	3.17	C D E
South	Midland	3.15	0.176	529	2.66	3.64	D E F
South	Coast	3.27	0.176	529	2.78	3.76	E F
Central	Midland	3.57	0.176	529	3.08	4.06	F

3. 4. Species abundance in relation to location

Species abundance, like species richness might also differ by location on a coast to mountain-gradient and a south to north gradient. A two-way analysis of variance tested the farmland bird species abundance on gradients ranging from coast to mountain (coast gradient) and south to north (north gradient). Species abundance showed a significant difference in the variable coast gradient ($F_{2,529}$= 36.03, p < 0.001, η^2 = .100), with an overall lower species abundance in the mountains (figure 5B). Species abundance was also consistently lower in the northern locations ($F_{2,529}$= 39.29, p < 0.001, η^2 = .109) (figure 5B). The interaction between the coast to mountain gradient and the south to north gradient was also significant ($F_{4,529}$= 10.23, p < 0.001, η^2 = .057): The location with the lowest mean abundance was in the northern part of the coast, while the location with the highest mean abundance was in the central midland (figure 5B).

Like the results for species richness, the main effects, and the interaction of the coast and the north gradient were significantly different. The post-hoc test of species abundance showed that the least square mean of the locations differed less between groups than in the post-hoc test for species richness (table 4 and 5), but differed greatly between the locations with the highest and lowest least square mean. Also, here the area with the highest least square mean was central midland (lsmean = 8.93) and the lowest least square mean in north coast (lsmean = 2.75).

Table 5. Post-hoc test of species abundance for the nine study locations, with least square means, standard errors, degrees of freedom, upper and lower confidence interval, and least square mean groups. Means sharing a letter are not significantly different.

North Gradient	Coast gradient	Lsmean	SE	Df	Lower.CL	Upper.CL	Group
North	Coast	2.75	0.462	529	1.47	4.03	A
North	Mountain	2.97	0.462	529	1.68	4.25	A B
South	Mountain	3.58	0.462	529	2.30	4.87	A B
North	Midland	4.25	0.462	529	2.97	5.53	A B
Central	Coast	4.55	0.462	529	3.27	5.83	A B
Central	Mountain	4.91	0.470	529	3.61	6.22	B
South	Coast	7.52	0.462	529	6.23	8.80	C
South	Midland	7.77	0.462	529	6.48	9.05	C
Central	Midland	8.93	0.462	529	7.65	10.22	C

3. 5. Landscape complexity in relation to location

The landscape complexity (fractal dimension index) was tested as an exploratory variable against the response variables coast to mountain gradient and south to north gradient in a two-way ANOVA. Landscape complexity was significantly different over the coast to mountain gradient ($F_{2,529}$= 13.07, p < 0.001, η^2 = .045), with the coast having an overall higher landscape complexity and the midlands having an overall high landscape complexity (figure 5C). The south to north gradient was also significantly different ($F_{2,529}$= 5.09, p = 0.006, η^2 = .017). In addition, the interaction between the two gradients were significantly different ($F_{4,529}$= 5.26, p < 0.001, η^2 = .036): The location with the highest mean fractal dimension index value was in the north mountains, while the lowest value was in the south mountains (figure 5C). The variability was highest between the three mountain areas, while the areas in midland and coast were clustered with relative similar complexity values.

The post-hoc test of landscape complexity (table 6) showed that the differences in landscape complexity were greatest between central midland (lsmean = 0.0185) and the south mountain (lsmean = 0.0252). The three mountain locations differed the most between each other, while the locations on the coast and midland were more like each other.

Table 6. Post-hoc test of landscape complexity for the nine study locations, with least square means, standard errors, degrees of freedom, upper and lower confidence interval, and least square mean groups. Means sharing a letter are not significantly different.

North Gradient	Coast gradient	Lsmean	SE	Df	Lower.CL	Upper.CL	Group
Central	Midland	0.0185	0.000894	529	0.0160	0.0210	A
North	Mountain	0.0185	0.000894	529	0.0161	0.0210	A
North	Midland	0.0201	0.000894	529	0.0176	0.0226	A B
South	Midland	0.0204	0.000894	529	0.0179	0.0229	A B
Central	Mountain	0.0222	0.000909	529	0.0197	0.0247	A B C
Central	Coast	0.0227	0.000894	529	0.0202	0.0252	B C
South	Coast	0.0235	0.000894	529	0.0210	0.0260	B C
North	Coast	0.0240	0.000894	529	0.0215	0.0265	B C
South	Mountain	0.0252	0.000894	529	0.0227	0.0277	C

3. 6. Landscape diversity in relation to location

Landscape diversity (Shannon index) was tested as an explanatory variable in a two-way ANOVA with the coast to mountain gradient and the south to north gradient as response variables. The coast to mountain gradient showed a significant difference in landscape diversity ($F_{2,529}$= 26.86, p < 0.001, η^2 = .091), with an overall high landscape diversity in the coastal locations (figure 5D). The landscape diversity over the south to north gradient was not significantly different ($F_{2,529}$= 2.57, p = 0.077, η^2 = .008). The interaction between the two gradients was not significantly different (p = 0.243): The north coast had the highest mean Shannon index value, and the central midland had the lowest mean value (figure 5D).

3. 7. Binary logistic regression of the eight common farmland species

Most of the farmland bird species showed a positive response to the proportion of fully cultivated soil. There were two species that showed a positive response to landscape diversity, while two species showed a negative effect from landscape complexity. The overall model fit (McFadden pseudo-R^2) of the logistic regression analysis ranged between values of 0.03 and 0.35, and most of the models showed a low decrease in AIC score from their corresponding null model after a stepwise procedure, indicating that most of the models had relatively a low fit.

The final model for predicting barn swallow presence in the survey plots (Appendix A1) had an AIC score of 657.7 opposed to the full model with a score of 685.03 (Likelihood ratio test, $\chi^2 = 37.29$, $p < 0.001$). The variables included in the final model were fully cultivated soil, Shannon index, fractal dimension index and marshes. Fully cultivated soil ($\beta = 2.67e^{-06}$, $\chi^2 = 18.93$, $p < 0.001$) and Shannon index ($\beta = 1.016$, $\chi^2 = 6.12$, $p = 0.013$) were the only variables that were significant and both had a positive coefficient, i.e. a higher probability of a barn swallow being present where there is a greater amount of fully cultivated soil but also a wide range of habitats. The model explained 5 % of the variance (McFadden $\rho^2 = 0.05$).

The final model for eurasian curlew (Appendix A2) had an AIC score of 510.04, the full model had a score of 552.41 (Likelihood ratio test, $\chi^2 = 52.04$, $p < 0.001$). The variables included in the final model were forest, surface cultivated soil, open land, infrastructure, and fractal dimension index. Forest ($\beta = -5.73e^{-06}$, $\chi^2 = 39.66$, $p < 0.001$), open land ($\beta = -2.29e^{-06}$, $\chi^2 = 5.93$, $p = 0.015$) and infrastructure ($\beta = -2.97e^{-05}$, $\chi^2 = 3.98$, $p = 0.046$) were the only variables that were significant. All the significant values had a negative coefficient, i.e. a lower probability of curlew being present where forest cover is higher. The model explained 9 % of the variance (McFadden $\rho^2 = 0.09$)

Common starling had a final model AIC score of 647.2 (Appendix A3) with a full-model AIC score at 747.22 (Likelihood ratio test, $\chi^2 = 37.29$, $p < 0.001$). The variables included in the final model were fully cultivated soil, ocean, Shannon index fresh water and marshes. The significant variables were fully cultivated soil ($\beta = 4.71e^{-06}$, $\chi^2 = 49.24$, $p < 0.001$), ocean ($\beta = 3.31e^{-06}$, $\chi^2 = 8.61$, $p = 0.003$), Shannon index ($\beta = 1.52$, $\chi^2 = 13.83$, $p < 0.001$) and fresh water ($\beta = -4.96e^{-06}$, $\chi^2 = 7.01$, $p = 0.008$). Fresh water had a negative coefficient while the other significant variables had positive coefficients. The model explained 14 % of the variance (McFadden $\rho^2 = 0.14$).

The northern lapwing had a final model AIC score of 241.1 (Appendix A4) with a full-model score of 260.95 (Likelihood ratio test, $\chi^2 = 25.86$, $p < 0.001$). Variables included in the final model were fully cultivated soil, built up area and open land. The only significant variable was fully cultivated soil ($\beta = 3.09e^{-06}$, $\chi^2 = 9.09$, $p = 0.003$) that had a positive coefficient. The model explained 10 % of the variance (McFadden $\rho^2 = 0.09$).

The final model for white wagtail had an AIC score of 671 (Appendix A5), where the full model had an AIC score of 682.18 (Likelihood ratio test, $\chi^2 = 21.20$, $p < 0.001$). The variables included in the final model were infrastructure, forest, pasture, Shannon index and fresh water. Infrastructure ($\beta = -3.61e^{-}$

^{05}, $\chi^2 = 7.40$, $p = 0.007$) and pasture ($\beta = -8.59e^{-06}$, $\chi^2 = 4.85$, $p = 0.028$) were the only significant variables. The model explained 3 % of the variance (McFadden $\rho^2 = 0.03$).

Skylark had a final model with an AIC score of 199.4 (Appendix A6), the full model had an AIC score of 296.85 (Likelihood ratio test, $\chi^2 = 103.42$, $p < 0.001$). The variables included in the final model were fully cultivated soil, fractal dimension index and open land. Fully cultivated soil ($\beta = 9.42e^{-06}$, $\chi^2 = 45.44$, $p < 0.001$), and fractal dimension index ($\beta = -9.98e^{+01}$, $\chi^2 = 6.83$, $p = 0.009$) were significant. Fully cultivated soil had a positive coefficient and fractal dimension index had a negative coefficient. The final model explained 35 % of the variance (McFadden $\rho^2 = 0.35$).

The final model of the yellowhammer had an AIC score of 451.2 (Appendix A7), and the full model had an AIC score of 674.75 (Likelihood ratio test, $\chi^2 = 229.53$, $p < 0.001$). Variables included in the final model were fully cultivated soil, fractal dimension index and infrastructure. Fully cultivated soil ($\beta = 1.12e^{-05}$, $\chi^2 = 76.37$, $p < 0.001$), fractal dimension index ($\beta = -7.56e^{+01}$, $\chi^2 = 21.30$, $p < 0.001$) and infrastructure ($\beta = 9.16e^{-05}$, $\chi^2 = 15.12$, $p < 0.001$) were all significant variables. Fully cultivated soil and infrastructure had positive coefficients and fractal dimension index had a negative coefficient. The model explained 34 % of the variance (McFadden $\rho^2 = 0.34$).

Whinchat had a final model AIC score of 635.9 (Appendix A8) and a full model AIC score of 665.28 (Likelihood ratio test, $\chi^2 = 41.38$, $p < 0.001$). The final model variables included pasture, infrastructure, fully cultivated soil, surface cultivated soil, Shannon index and open. The variables that were significant were pasture ($\beta = 1.06e^{-05}$, $\chi^2 = 7.98$, $p = 0.005$), infrastructure ($\beta = -3.97e^{-05}$, $\chi^2 = 6.22$, $p = 0.013$) and fully cultivated soil ($\beta = 2.36e^{-06}$, $\chi^2 = 12.10$, $p < 0.001$). Pasture and fully cultivated soil had positive coefficients, infrastructure had a negative coefficient. The model explained 6 % of the variance (McFadden $\rho^2 = 0.06$).

4. Discussion

This study aimed to investigate the habitat preference and distribution of eight common farmland bird species in central Norway. The study showed that the midlands and southern parts of the coastal locations had the greatest species richness and abundance of farmland birds. The study also showed that most of the farmland birds had a positive response to agricultural habitats such as fully cultivated soil.

Predicting patterns in nature is a difficult task and there are a lot of different factors that could influence the outcome. Inclusion of such factors in analyses is therefore a key to the understanding of species' ecology and behaviour. In this study the use of landcover variables were restricted to the variables presented by NIBIO classification system FKB-AR5 (table 1), and indices calculated from those variables. The usage of these variables was due to the availability of landcover data, and therefore the possibility to implement these variables in a GIS-analysis. The variables from the FKB-AR5 classification system only describe eleven main landcover types, this classification system might not be detailed enough to include all the habitat features influencing the presence of the farmland bird species. For example, several studies describing farmland birds refer to habitat types that are not differentiated in the variables in the analysis, such as hedgerows (Evans *et al.*, 2007; McHugh *et al.*, 2018), wetlands, rough grasslands, and moorland (Brown *et al.*, 2015; Grant, 1997). Other studies have used different farmland types such as grassland and different kinds of cereal as variables in their analyses (Evans *et al.*, 2007; Valkama *et al.*, 1998). To supplement the somewhat broad landcover variables used in the analyses, variables for landscape diversity and landscape complexity were included.

Since there are only a few studies on the distribution and habitat preferences of farmland bird species in Norway, we also chose to examine and compare our results with studies of farmland birds in other European countries. Farmland in western Europe has for a longer time been managed more intensively than in Norway, and the negative effects of intensification are also more prevalent for farmland bird populations in these countries. Norwegian farmland constitutes only 3.5 % of the total land area (SSB, 2019). In most of the European countries the percentages of farmland area are much larger, for example, in the EU countries farmlands make up 48 % of the total land area (BirdLife, 2019).

The eight farmland bird species that were included as indicator species in this study were based on the suggestions given by Husby and Kålås (2011). They based their selection on various criteria, where the two main criteria were: i) That a major portion of the population of a species are found in a particular habitat during the nesting season. ii) That there are satisfactory quantitative population data of the species from the survey project TOV-E. Siddig *et al.* (2016) remarks upon the increased use of indicator species in the past 20 years. The use of indicator species for monitoring environmental changes is reliable and cost-effective, but Siddig *et al.* (2016) sheds light on the challenges of selecting specific indicators and identification of the relationship between these indicators and their specific application. In their review, they conclude that the future use of indicator species depend on "rigorously selected groups of indicators that reflect the environment in realistic ways and also reflect cause-effect relationships between the indicator species and underlying processes of interest". Siddig *et al.* (2016) suggests a five-step process on which indicator species should be selected and used in monitoring environmental changes: 1) Set clear monitoring goals that can be reflected by the selected indicator species. 2) Identify the ecological setting and spatial extent of the study site. 3) Selecting the candidate indicator species and demographic parameters based on criteria given by Cairns and Pratt (1993), Dale and Beyeler (2001) and Carignan and Villard (2002). 4) Select ecological covariates/predictors to which the indicator species is particularly responsive. 5) Sample species abundance and ecosystem covariates simultaneously, then conduct an indicator species analysis to get the indicator value for each species following the method of Dufrêne and Legendre (1997). On basis of these criteria, the selection criteria of Husby and Kålås (2011) is not far from the suggestions of Siddig *et al.* (2016). In this case, the goal of the monitoring project is to monitor the populations of farmland birds using eight common farmland birds as indicators. These indicators where selected based on their preference of agricultural habitat over other habitats during the nesting season. Choosing the right indicator species is an important part of ecological research and these steps may prove important in selecting species as indicator for environmental changes in the future.

The recordings of the survey showed that at least one of the eight common farmland bird species were present on 89 % of all plots combined throughout the years 2013, 2014 and 2019, and that both the species richness and abundance had a slight decrease between the year 2014 and 2019. Although this negative population trend is supported by other studies written on farmland bird species (Donald *et al.*, 2001; Inger *et al.*, 2015), three years of monitoring is a too short time frame to identify population trends.

The descriptive analysis of species presence per location (figure 3) and landcover distribution (figure 4) showed that the locations which were more dominated by fully cultivated soil, such as the central and southern parts of the midland and coastal, had a higher presence per plot of the eight species than in the other locations to the north and to the mountains. The result from these analyses could further support the research by Husby and Kålås (2011) on selecting farmland bird indicators for

Norway, as these results seem to correspond with their findings that the farmland birds species prefer agricultural locations.

4. 1. Species- richness and abundance, and landscape- complexity and diversity in relation to location

The analyses of species richness and species abundance, and landscape complexity and landscape diversity, in relation to location were included in the study to further describe the landscape features based on their location in the study area. These results suggest that species richness and species abundance of the eight common farmland bird species are generally higher in the central and southern parts of the coast and midland. The results for landscape complexity and landscape diversity (figure 5) showed a pattern where the coastal locations had generally higher complexity and diversity, while the locations in the midlands had generally lower landscape complexity and diversity. The locations in the mountain varied in complexity from a south to north gradient and had an overall low landscape diversity.

There seems to be a link between the findings in figure 4 and the findings of the species richness and species abundance analyses (figure 5A and 5B). The locations with high proportions of fully cultivated soil found in the midlands and the southern parts of the coast also had a higher mean species richness and species abundance. These results also correspond with the findings of Pedersen and Krøgli (2017) showing a positive correlation between species richness and species abundance of farmland birds and the amount of agricultural area in Norway. (Pedersen & Krøgli, 2017) also found a strong significance between richness and abundance of farmland birds and landscape heterogeneity, which in turn could support our findings regarding the higher species richness and abundance found in the central and southern parts of the coastal area (figure 5C and 5D). The locations in the mountains had the highest proportion of forest cover (figure 4) and the lowest richness and abundance of farmland bird species (Figure 5A and 5B). This might suggest that forest could be a limiting factor for richness and abundance of farmland bird species. This is supported by Berg *et al.* (2015) who found that forest had a negative effect on the abundance of some farmland bird species. They also found that landscape heterogeneity had a positive effect on occurrence of several non-crop nesting species, indicating that non-crop elements such as forest edges, habitat islands and farmsteads contributes to an overall landscape heterogeneity that influence the occurrence of these species. Pickett and Siriwardena (2011) found that while landscape heterogeneity might be beneficial for some species associated with farmland, it may negatively impact others such as northern lapwing and eurasian skylark, suggesting an avoidance of such habitats. The results of that study agree with the results of this study (figure 3 and figure 5C and 5D) where the more complex and diverse landscapes at the northern coast showed an overall lower presence of northern lapwing and eurasian skylark.

4. 2. Single species analysis

The logistic regression analysis of the presence of the eight common farmland bird species demonstrated that most of the species showed a significant positive effect of increased cover of farmland habitat types. In the final models, six out of eight species had a positive effect from either fully cultivated soil, surface cultivated soil and pasture. The species which did not have a clear relationship with farmland habitats were eurasian curlew and white wagtail. Eurasian curlew showed a slight non-significant response to surface cultivated soil, while white wagtail presence showed a negative effect from pastures. The results of the logistic regression analyses of eight farmland bird species may correspond with earlier research on selecting indicator species for farmland in Norway, as well as in some European countries (Engan *et al.*, 2008; Husby & Kålås, 2011; Kålås *et al.*, 2016).

Most of the final models showed low fit based on their AIC score and McFadden pseudo-R^2 value. The models that had the best fit were the models for skylark (ρ^2 = 0.35) and yellowhammer (ρ^2 = 0.34), while the model with the lowest pseudo-R^2 was the model for white wagtail (ρ^2 = 0.03). As most of the final models had a low fit it is important to keep in mind that the models might not have that much power in predicting the outcomes of presence based on the variables included in the final models. An overall low fit of the models could indicate that there were other variables that could have predicted the presence of the farmland bird species in a more accurate way than the ones included in the analysis, or that the variables included in the models had a strong collinearity (Graham, 2003). Including variables describing food availability could have improved the model fit in the analyses, for example variables for the densities of invertebrates or variables describing different crop types. Inclusion of an invertebrate density variable would rely on the availability of such data, if such data was unobtainable externally one would have to collect the data oneself, which could ensue in higher cost and time used on the project. The same is the case for data on crop types, but these data are probably available from institutions such as the Norwegian Map Authority. When using variables of landcovers it is a chance that there will occur collinearity in the analysis. Collinearity might be a reason for low fit in the final models, as there are up to ten different landcover types included in the analyses. The coast to mountain gradient and the south to north gradient were not included in the logistic analyses as landcover and the gradients might be related (figure 4) and would by this add to the collinearity that may already be present in the models.

The landcover types that had the most positive response on presence of farmland birds in the models were fully cultivated soil and pastures. This might correspond with the findings in figure 4, where the locations with the most agricultural landcover types were found in the midlands and the two southern locations at the coast. These locations also had the highest presence of farmland birds (figure 3).

Fully cultivated soil and landscape diversity had a positive effect on the presence of barn swallow. The habitat preference of barn swallow is tied to its foraging habits as an insectivore catching flying insects. The positive effect from fully cultivated soil could be related to the fact that barn swallows prefer to hunt insects in open terrain such as pastures and farmland (Musitelli *et al.*, 2016) and that these habitat types to a larger degree are inhabited by flying insects. An explanation for the positive effect of landscape diversity could be that barn swallow benefit from the presence of field margins in its habitat, as these field margins also might increase the number of flying invertebrates in an area. Other research supports this fact and emphasize the importance of field margins and hedgerows for the barn swallows (Evans *et al.*, 2007; McHugh *et al.*, 2018). What differed from our studies was that Evans *et al.* (2007) found the densities of barn swallow to be two to four times higher on pastures than over other types of crop fields. One reason for this could be the small proportions of pastures we encountered in our study area compared to area of fully cultivated soil.

In the final model for the Eurasian curlew we found that habitat types such as forest, infrastructure, and open land had a negative effect on the species presence. The negative effect of forest habitat might be due to the presence of predators and their effect on breeding success of the species. The eurasian curlew nests on the ground, making the nests particularly vulnerable to predation. Several studies argue that predators could be one of the key drivers in population decline of eurasian curlew in Europe, and that predators are linked to the presence of forest. For example, forests have “edge” effects well beyond their boundaries by supporting populations of predators, which increases the predation pressure in bordering habitats (Berg, 1992; Brown *et al.*, 2015; Douglas *et al.*, 2014; Valkama & Currie, 1999; Wilson *et al.*, 2014). One reason that we did not find any habitat types that had a positive effect on the presence of eurasian curlew in its final model could be that curlew also prefer to breed on margins of farmland such as wetlands, rough grasslands and moorland, and avoids short grasslands with tillage (Grant, 1997; Valkama *et al.*, 1998). These habitat types were not variables categorised in the analysis of the landscape in the study area and have might been mixed in with other habitat variables surrounding the farmland.

The final model for common starling showed a positive effect of fully cultivated soil, ocean, landscape diversity and a negative effect from fresh water. That fully cultivated soil had a positive effect coincides with other studies where it is shown that common starling prefer farmland habitats (Gregory & Baillie, 1998). Gregory and Baillie (1998) also claim that common starling is closely related to human activity, while this did not appear in the final model with non-agricultural human-made habitats such as built-up area and infrastructure, the reason for this could come of that the study locations and the plots were mostly in agricultural landscapes and to only a small degree located in urban areas. The positive effect from ocean in the model could be caused by the fact that the locations where there was a high presence of common starling, also was in relation to the sea and the fjord in the study area, and probably also in relation to human activity, or it could also just be a coincident.

The final model for northern lapwing showed a positive effect of fully cultivated soil. This is also supported by Wilson *et al.* (2001), who found high numbers of northern lapwing in relation to grasslands and arable habitats. Husby and Kålås (2011) found that 80 % of the northern lapwing in their study bred in agricultural landscape, and the remaining 20 % in wetlands. These results also fit well with the findings in figure 3 where a good proportion of northern lapwing in the study were found in the agriculture dominated locations such as Stjørdal and Levanger/Verdal.

Infrastructure and pasture were both variables that had a negative effect on the presence of white wagtail in the final model. This final model had the lowest McFadden ρ^2 of all the models, which could be the reason for the final model ending up with significant effects of variables that were not expected. White wagtail was included as a farmland bird species indicators in the research by Husby and Kålås (2011), where they are described as preferring farmland habitats mainly fully cultivated soil and pastures. These claims differ from our findings where the final model for white wagtail showed a significant negative effect of pastures. A reason for these conflicting findings could come of the weak model fit, and that there might were some other variables that could have described the presence of white wagtail in a better way, or that the species could be more versatile in its habitat choice than the other species.

The skylark is known to be a common bird found in the farmland fields, which reflects with the results of the final model in the study. In the final model skylark showed a clear positive significance of fully cultivated soil, and a clear negative effect of complex landscapes. This correspond well with other findings, as the species is well documented to be associated with agricultural landscape (Gregory & Baillie, 1998; Szilassi *et al.*, 2019; Toepfer & Stubbe, 2001). Also, Robinson *et al.* (2001) found that with increased area of arable land, the presence of skylark increased as well. Wilson *et al.* (1997) found that skylark had a higher breeding success on set-aside farmland rather than intensively managed farmland and that the species benefits from mixed farming. The benefits skylark gain from set-aside land is also emphasized by (Toepfer & Stubbe, 2001). As of today, the Norwegian agricultural practice is not that intensively managed as in the rest of Europe, which in turn could be beneficial for its presence in this habitat type and the reason for such strong significance in the model. The skylark was also one of the final models with the highest predicting power ($\rho^2 = 0.35$), a contributing factor for the relative high pseudo-R^2 value could be that skylark require large open areas, without tall vegetation or other structures, making the structure of field vegetation important in skylark habitat preference (Piha *et al.*, 2003).

Yellowhammer was the most common farmland bird species in the study. The final model showed that yellowhammer had a clear positive effect of fully cultivated soil, a slight positive effect from infrastructure and a negative effect of landscape complexity. That yellowhammer has a positive effect from fully cultivated soil corresponds with other findings, e.g. Robinson *et al.* (2001), describes that yellowhammer increased with the amount of arable land. This claim may also correspond with the negative effect of landscape complexity in the model since the amount of farmland might contradict the effect of complexity in a habitat. This may also be a reason why the final model for yellowhammer got a relatively high predicting power ($\rho^2 = 0.34$).

The final model of the whinchat had a positive effect of both pastures and fully cultivated soil. This correspond with the findings of Fischer *et al.* (2013), who found that whinchat territories were characterized by high proportions of pastures and grassland. The positive effect that whinchat has to pasture and fully cultivated soil could be because whinchats are ground nesting and prefer landscapes with perches, which habitat types such as pastures and grassland may have more of than the other landscape types in the model. One should also keep in mind that the final model of whinchat also has one of the lowest model-fits in the analysis ($\rho^2 = 0.06$), reducing its predicting power of its final model.

The findings of this study might support the decision of using the selected eight bird species as indicators for habitat quality in Norway. Using indicator species for monitoring ecological qualities has been proven to be an applicable and cost-effective method. However it could be wise to implement a set of guidelines when selecting indicators species in the future, for example, by following the suggestions of Siddig *et al.* (2016). Furthermore, this study could contribute on emphasizing the importance of suitable habitats, landscape diversity and landscape complexity, and its effects on farmland bird species richness and abundance in a country in northern Europe. The final models of the regression analyses had mostly low predicting capabilities, suggesting collinearity between the variables, or a too “narrow” choice of variables. Although some of the models showed a low model fit, most of the final model results showed what could be expected as habitat preferences for the farmland bird species.

References

Agresti, A. (2003). *Categorical data analysis* (Vol. 482): John Wiley & Sons.

Ahlstrøm, A., Bjørkelo, K., & Fadnes, K. D. (2019). AR5 Klassifikasjonssystem. *NIBIO Bok*.

Akaike, H. (1974). A new look at the statistical model identification. *IEEE transactions on automatic control, 19*(6), 716-723.

Anderson, S. H., Kelly, D., Ladley, J. J., Molloy, S., & Terry, J. (2011). Cascading effects of bird functional extinction reduce pollination and plant density. *Science, 331*(6020), 1068-1071.

Batáry, P., Matthiesen, T., & Tscharntke, T. (2010). Landscape-moderated importance of hedges in conserving farmland bird diversity of organic vs. conventional croplands and grasslands. *Biological Conservation, 143*(9), 2020-2027.

Bennett, A. F., Nimmo, D. G., & Radford, J. Q. (2014). Riparian vegetation has disproportionate benefits for landscape-scale conservation of woodland birds in highly modified environments. *Journal of Applied Ecology, 51*(2), 514-523.

Berg, Å. (1992). Factors affecting nest-site choice and reproductive success of Curlews Numenius arquata on farmland. *Ibis, 134*(1), 44-51.

Berg, Å., Wretenberg, J., Żmihorski, M., Hiron, M., & Pärt, T. (2015). Linking occurrence and changes in local abundance of farmland bird species to landscape composition and land-use changes. *Agriculture, Ecosystems & Environment, 204*, 1-7. doi:10.1016/j.agee.2014.11.019

BirdLife. (2019). The EU Biodiversity Strategy to 2030. Retrieved from https://www.birdlife.org/sites/default/files/attachments/birdlife_position_biodiversity_web_01.pdf

BirdLife, F. v. B. (2004). *Birds in Europe: population estimates, trends and conservation status*: Cambridge.

Bradbury, R. B., Kyrkos, A., Morris, A. J., Clark, S. C., Perkins, A. J., & Wilson, J. D. (2000). Habitat associations and breeding success of yellowhammers on lowland farmland. *Journal of Applied Ecology, 37*(5), 789-805.

Brown, D., Wilson, J., Douglas, D., Thompson, P., Foster, S., McCulloch, N., . . . Crockford, N. (2015). The Eurasian Curlew–the most pressing bird conservation priority in the UK. *British birds, 108*, 660-668.

Burnham, K. P., Anderson, D. R., & Huyvaert, K. P. (2011). AIC model selection and multimodel inference in behavioral ecology: some background, observations, and comparisons. *Behavioral ecology and sociobiology, 65*(1), 23-35.

Cairns, J., & Pratt, J. R. (1993). A history of biological monitoring using benthic macroinvertebrates. *Freshwater biomonitoring and benthic macroinvertebrates, 10*, 27.

Carignan, V., & Villard, M.-A. (2002). Selecting indicator species to monitor ecological integrity: a review. *Environmental monitoring and assessment, 78*(1), 45-61.

Colwell, R. K. (2009). Biodiversity: concepts, patterns, and measurement. *The Princeton guide to ecology, 663*, 257-263.

Dale, S., & Hardeng, G. (2016). Changes in the breeding bird communities on mires and in surrounding forests in southeastern Norway during a 40-year period (1976–2015). *Ornis Norvegica, 39*, 11-24.

Dale, V. H., & Beyeler, S. C. (2001). Challenges in the development and use of ecological indicators. *Ecological Indicators, 1*(1), 3-10.

Dallimer, M., Skinner, A. M., Davies, Z. G., Armsworth, P. R., & Gaston, K. J. (2012). Multiple habitat associations: the role of offsite habitat in determining onsite avian density and species richness. *Ecography, 35*(2), 134-145.

Donald, P. F., Gree, R. E., & Heath, M. F. (2001). Agricultural intensification and the collapse of Europe's farmland bird populations. *Proc Biol Sci, 268*(1462), 25-29. doi:10.1098/rspb.2000.1325

Donald, P. F., Sanderson, F. J., Burfield, I. J., & van Bommel, F. P. J. (2006). Further evidence of continent-wide impacts of agricultural intensification on European farmland birds, 1990–2000. *Agriculture, Ecosystems & Environment, 116*(3-4), 189-196. doi:10.1016/j.agee.2006.02.007

Douglas, D. J., Bellamy, P. E., Stephen, L. S., Pearce–Higgins, J. W., Wilson, J. D., & Grant, M. C. (2014). Upland land use predicts population decline in a globally near-threatened wader. *Journal of Applied Ecology, 51*(1), 194-203.

Dufrêne, M., & Legendre, P. (1997). Species assemblages and indicator species: the need for a flexible asymmetrical approach. *Ecological monographs, 67*(3), 345-366.

Engan, G., Bratli, H., Fjellstad, W. J., & Dramstad, W. (2008). 3Q-Biologisk mangfold i jordbrukets kulturlandskap. Status og utviklingstrekk.

ESRI. (2019). ArcGis Pro (Version 2.5): Environmental Systems Research Institute

Evans, K. L., Wilson, J. D., & Bradbury, R. B. (2007). Effects of crop type and aerial invertebrate abundance on foraging barn swallows Hirundo rustica. *Agriculture, Ecosystems & Environment, 122*(2), 267-273.

Fischer, K., Busch, R., Fahl, G., Kunz, M., & Knopf, M. (2013). Habitat preferences and breeding success of Whinchats (Saxicola rubetra) in the Westerwald mountain range. *Journal of ornithology, 154*(2), 339-349.

Fox, A. (2004). Has Danish agriculture maintained farmland bird populations? *Journal of Applied Ecology, 41*(3), 427-439.

Geiger, F., Bengtsson, J., Berendse, F., Weisser, W. W., Emmerson, M., Morales, M. B., . . . Winqvist, C. (2010). Persistent negative effects of pesticides on biodiversity and biological control potential on European farmland. *Basic and Applied Ecology, 11*(2), 97-105.

Gelman, A., & Hill, J. (2006). *Data analysis using regression and multilevel/hierarchical models*: Cambridge university press.

Gillings, S., Henderson, I. G., Morris, A. J., & Vickery, J. A. (2010). Assessing the implications of the loss of set-aside for farmland birds. *Ibis, 152*(4), 713-723.

Gjershaug, J. O. (1994). *Norsk fugleatlas: hekkefuglenes utbredelse og bestandsstatus i Norge*: Norsk ornitologisk forening.

Graham, M. H. (2003). Confronting multicollinearity in ecological multiple regression. *Ecology, 84*(11), 2809-2815.

Grant, M. (1997). Breeding curlew in the UK: RSPB research and implications for conservation. *RSPB Conservation Review, 11*, 67-73.

Green, M., Lindström, Å., & Haas, F. (2016). *Övervakning av fåglarnas populationsutveckling: årsrapport för 2013*: Biologiska institutionen, Lunds universitet.

Gregory, R. D., & Baillie, S. (1998). Large-scale habitat use of some declining British birds. *Journal of Applied Ecology, 35*(5), 785-799.

Gregory, R. D., & Van Strien, A. (2010). Wild bird indicators: using composite population trends of birds as measures of environmental health. *Ornithological Science, 9*(1), 3-22.

Gregory, R. D., Van Strien, A., Vorisek, P., Gmelig Meyling, A. W., Noble, D. G., Foppen, R. P., & Gibbons, D. W. (2005). Developing indicators for European birds. *Philosophical Transactions of the Royal Society B: Biological Sciences, 360*(1454), 269-288.

Heggøy, O., & Øien, I. (2014). Vipa går en usikker framtid i møte. *Vår Fuglefauna, 37*, 114-127.

Heikkinen, R. K., Luoto, M., Virkkala, R., & Rainio, K. (2004). Effects of habitat cover, landscape structure and spatial variables on the abundance of birds in an agricultural–forest mosaic. *Journal of Applied Ecology, 41*(5), 824-835.

Henderson, I. G., Holland, J. M., Storkey, J., Lutman, P., Orson, J., & Simper, J. (2012). Effects of the proportion and spatial arrangement of un-cropped land on breeding bird abundance in arable rotations. *Journal of Applied Ecology, 49*(4), 883-891.

Henriksen, S., & Hilmo, O. (2015). Norsk rødliste for arter 2015. *Artsdatabanken, Norge, 6*.

Hilty, J., & Merenlender, A. (2000). Faunal indicator taxa selection for monitoring ecosystem health. *Biological Conservation, 92*(2), 185-197.

Hosmer Jr, D. W., Lemeshow, S., & Sturdivant, R. X. (2013). *Applied logistic regression* (Vol. 398): John Wiley & Sons.

Husby, M., & Kålås, J. A. (2011). Terrestriske fuglearter som indikatorer for bærekraftig utvikling i Norge: tilstanden i ulike naturtyper og effekter av klimaendring.

Husby, M., & Reinsborg, T. (2015). Ornitologisk rapport fra Eidsbotn og Levangersundet, med bestandsendringer fra 1996 til 2015.

Inger, R., Gregory, R., Duffy, J. P., Stott, I., Voříšek, P., & Gaston, K. J. (2015). Common European birds are declining rapidly while less abundant species' numbers are rising. *Ecology letters, 18*(1), 28-36.

Krummel, J., Gardner, R., Sugihara, G., O'neill, R., & Coleman, P. (1987). Landscape patterns in a disturbed environment. *Oikos*, 321-324.

Kålås, J. A., & Byrkjedal, I. (1981). *Vadefuglenes hekkestatus i Norge med Svalbard.* Paper presented at the Proceedings of the Second Nordic Congress of Ornithology.

Kålås, J. A., & Husby, M. (2002). Ekstensiv overvåking av terrestre fugl i Norge. *NINA-Oppdragsmelding, 740*, 25.

Kålås, J. A., & Husby, M. (2011). Det nye nasjonale nettverket for overvåking av terrestriske hekkefugler er nå etablert. *Vår Fuglefauna, 34*, 16-19.

Kålås, J. A., Husby, M., Nilsen, E., & Vang, R. (2014). Bestandsvariasjoner for terrestriske fugler i Norge 1996-2013.

Kålås, J. A., Husby, M., & Vang, R. (2016). TOV-E–artsindekser på nett. *Vår Fuglefauna, 39*, 24-28.

Landres, P. B., Verner, J., & Thomas, J. W. (1988). Ecological uses of vertebrate indicator species: a critique. *Conservation Biology, 2*(4), 316-328.

Long, J. S., & Freese, J. (2006). *Regression models for categorical dependent variables using Stata*: Stata press.

Lovejoy, S. (1982). Area-perimeter relation for rain and cloud areas. *Science, 216*(4542), 185-187.

MacKenzie, D. I., Nichols, J. D., Royle, J. A., Pollock, K. H., Bailey, L., & Hines, J. E. (2017). *Occupancy estimation and modeling: inferring patterns and dynamics of species occurrence*: Elsevier.

Magurran, A. E. (1988). *Ecological diversity and its measurement*: Princeton university press.

Mandelbrot, B. B. (1983). *The fractal geometry of nature* (Vol. 173): WH freeman New York.

McHugh, N. M., Bown, B. L., & Clark, J. E. (2018). Use of field margins managed under an agri-environment scheme by foraging Barn Swallows Hirundo rustica. *Bird Study, 65*(3), 329-337.

Moen, A. (1998). *Vegetasjon*. [Hønefoss]: Norges geografiske oppmåling.

Musitelli, F., Romano, A., Møller, A. P., & Ambrosini, R. (2016). Effects of livestock farming on birds of rural areas in Europe. *Biodiversity and Conservation, 25*(4), 615-631.

O'Neill, R., Krummel, J., Gardner, R. e. a., Sugihara, G., Jackson, B., DeAngelis, D., . . . Christensen, S. (1988). Indices of landscape pattern. *Landscape ecology, 1*(3), 153-162.

Pedersen, C., & Krøgli, S. O. (2017). The effect of land type diversity and spatial heterogeneity on farmland birds in Norway. *Ecological Indicators, 75*, 155-163. doi:10.1016/j.ecolind.2016.12.030

Pickett, S. R., & Siriwardena, G. M. (2011). The relationship between multi-scale habitat heterogeneity and farmland bird abundance. *Ecography, 34*(6), 955-969.

Piha, M., Pakkala, T., & Tiainen, J. (2003). Habitat preferences of the Skylark Alauda arvensis in southern Finland. *Ornis Fennica, 80*(3), 97-110.

Post, E., & Forchhammer, M. C. (2004). Spatial synchrony of local populations has increased in association with the recent Northern Hemisphere climate trend. *Proceedings of the National Academy of Sciences, 101*(25), 9286-9290.

Preston, F. W. (1948). The commonness, and rarity, of species. *Ecology, 29*(3), 254-283.

Queen, J. P., Quinn, G. P., & Keough, M. J. (2002). *Experimental design and data analysis for biologists*: Cambridge University Press.

R Core Team. (2019). R: A Language and Environment for Statistical Computing: R Foundation for Statistical Computing. Retrieved from https://www.R-project.org/

Robinson, R. A., Wilson, J. D., & Crick, H. Q. (2001). The importance of arable habitat for farmland birds in grassland landscapes. *Journal of Applied Ecology, 38*(5), 1059-1069.

Root, T. L., Price, J. T., Hall, K. R., Schneider, S. H., Rosenzweig, C., & Pounds, J. A. (2003). Fingerprints of global warming on wild animals and plants. *Nature, 421*(6918), 57-60.

Şekercioğlu, Ç. H., Daily, G. C., & Ehrlich, P. R. (2004). Ecosystem consequences of bird declines. *Proceedings of the National Academy of Sciences, 101*(52), 18042-18047.

Shannon, C. E. (1948). A mathematical theory of communication. *Bell system technical journal, 27*(3), 379-423.

Shimmings, P., & Øien, I. (2015). Bestandsestimater for norske hekkefugler. *NOF-rapport 2015, 2*, 268.

Siddig, A. A., Ellison, A. M., Ochs, A., Villar-Leeman, C., & Lau, M. K. (2016). How do ecologists select and use indicator species to monitor ecological change? Insights from 14 years of publication in Ecological Indicators. *Ecological Indicators, 60*, 223-230.

Siriwardena, G. M., Baillie, S. R., Buckland, S. T., Fewster, R. M., Marchant, J. H., & Wilson, J. D. (1998). Trends in the abundance of farmland birds: a quantitative comparison of smoothed Common Birds Census indices. *Journal of Applied Ecology, 35*(1), 24-43.

Solheim, R. (1993). Endringer i kulturlandskapet-Betydningen for pattedyr og fugl. *Fylkesmannen i Hedmark, Miljøvernavdelingen, Rapport, 8*, 93.

SSB. (2019). Arealbruk og arealressurser. Retrieved from https://www.ssb.no/arealstat

Szilassi, P., Csikos, N., Galle, R., & Szep, T. (2019). Recent and Predicted Changes in Habitat of the Eurasian Skylark Alauda arvensis Based on the Link between the Land Cover and the Field Survey Based Abundance Data. *Acta Ornithologica, 54*(1), 59-71.

Thingstad, P. G., & Vie, G. E. (1995). Fugl som indikatorgruppe for miljøriktig utvikling av kulturlandskapet. Et forstudie av fuglefaunaen ved Mære landbruksskole.

Toepfer, S., & Stubbe, M. (2001). Territory density of the Skylark (Alauda arvensis) in relation to field vegetation in central Germany. *Journal für Ornithologie, 142*(2), 184-194.

Tukey, J. W. (1949). Comparing individual means in the analysis of variance. *Biometrics*, 99-114.

Valkama, J., & Currie, D. (1999). Low productivity of curlews Numenius arquata on farmland in southern Finland: causes and consequences. *Ornis Fennica, 76*(2), 65-70.

Valkama, J., Robertson, P., & Currie, D. (1998). *Habitat selection by breeding curlews (Numenius arquata) on farmland: the importance of grassland.* Paper presented at the Annales Zoologici Fennici.

Walther, G.-R., Post, E., Convey, P., Menzel, A., Parmesan, C., Beebee, T. J., . . . Bairlein, F. (2002). Ecological responses to recent climate change. *Nature, 416*(6879), 389.

Wenny, D. G., Devault, T. L., Johnson, M. D., Kelly, D., Sekercioglu, C. H., Tomback, D. F., & Whelan, C. J. (2011). The need to quantify ecosystem services provided by birds. *The auk, 128*(1), 1-14.

Whelan, C. J., Wenny, D. G., & Marquis, R. J. (2008). Ecosystem services provided by birds. *Annals of the New York academy of sciences, 1134*(1), 25-60.

Wilson, A. M., Vickery, J. A., & Browne, S. J. (2001). Numbers and distribution of Northern Lapwings Vanellus vanellus breeding in England and Wales in 1998. *Bird Study, 48*(1), 2-17.

Wilson, J. D., Anderson, R., Bailey, S., Chetcuti, J., Cowie, N. R., Hancock, M. H., . . . Thompson, D. B. (2014). Modelling edge effects of mature forest plantations on peatland waders informs landscape-scale conservation. *Journal of Applied Ecology, 51*(1), 204-213.

Wilson, J. D., Evans, J., Browne, S. J., & King, J. R. (1997). Territory distribution and breeding success of skylarks Alauda arvensis on organic and intensive farmland in southern England. *Journal of Applied Ecology*, 1462-1478.

Winqvist, C., Bengtsson, J., Aavik, T., Berendse, F., Clement, L. W., Eggers, S., . . . Liira, J. (2011). Mixed effects of organic farming and landscape complexity on farmland biodiversity and biological control potential across Europe. *Journal of Applied Ecology, 48*(3), 570-579.

Woodhouse, S., Good, J., Lovett, A., Fuller, R., & Dolman, P. (2005). Effects of land-use and agricultural management on birds of marginal farmland: a case study in the Llŷn peninsula, Wales. *Agriculture, Ecosystems & Environment, 107*(4), 331-340.

Wretenberg, J., Lindström, Å., Svensson, S., & Pärt, T. (2007). Linking agricultural policies to population trends of Swedish farmland birds in different agricultural regions. *Journal of Applied Ecology, 44*(5), 933-941.

Wretenberg, J., Lindström, Å., Svensson, S., Thierfelder, T., & Pärt, T. (2006). Population trends of farmland birds in Sweden and England: similar trends but different patterns of agricultural intensification. *Journal of Applied Ecology, 43*(6), 1110-1120.

Wretenberg, J., Pärt, T., & Berg, Å. (2010). Changes in local species richness of farmland birds in relation to land-use changes and landscape structure. *Biological Conservation, 143*(2), 375-381.

Appendices

Appendix A

Appendix A1.

Final Logistic regression models of barn swallow farmland bird species

Barn swallow~ Predictor	**Direction of effect**	**β**	**SE β**	**Wald's χ^2**	***df***	***p***
Fully cultivated soil	↑	2.673e-06	6.143e-07	18.9344	1	**1.35e-05**
Shannon Index	↑	1.016e+00	4.109e-01	6.1182	1	**0.013380**
Open land	↓	-1.517e-06	8.715e-07	3.0286	1	0.081810
Fractal dimension Index	↓	-2.815e+01	1.451e+01	3.7651	1	0.052333
Marshes	↑	2.473e-06	1.683e-06	2.1589	1	0.141749
Test				χ^2	***df***	***p***
Overall model evaluation						
Likelihood ratio test				37.287	5	**5.2464e-07**

Note. R programming: [package:" rcompanion","car","lmtest"]
McFadden R^2 = 0.0545907
Model Null AIC = 685.03
Model Final AIC = 657.7

Appendix A2.

Final Logistic regression model of eurasian curlew

Eurasian curlew~ Predictor	**Direction of effect**	**β**	**SE β**	**Wald's χ^2**	***df***	***p***
Forest	↓	-5.726e-06	9.092e-07	39.6646	1	**3.02e-10**
Surface cultivated soil	↑	2.687e-05	1.579e-05	2.8973	1	0.0887
Open land	↓	-2.287e-06	9.393e-07	5.9275	1	**.0149**
Infrastructure	↓	-2.973e-05	1.490e-05	3.9826	1	**.0460**
Fractal value	↑	2.461e+01	1.653e+01	2.2165	1	.1365
Test				χ^2	***df***	***p***
Overall model evaluation						
Likelihood ratio test				52.041	5	**5.291e-10**

Note. R programming: [package:" rcompanion","car","lmtest"]
McFadden R^2 = 0.0945495
Model Null AIC = 552.41
Model Final AIC = 510.4

Appendix A3.

Final Logistic regression model of common starling						
Common starling~ Predictor	**Direction of effect**	**β**	**SE β**	**Wald's χ^2**	***df***	***p***
Fully cultivated soil	↑	4.713e-06	6.716e-07	49.2375	1	**2.27e-12**
Ocean	↑	3.305e-06	1.126e-06	8.6110	1	**0.003341**
Shannon index	↑	1.522e+00	4.092e-01	13.8359	1	**0.000199**
Freshwater	↓	-4.956e-06	1.871e-06	7.0145	1	**0.008085**
Marshes	↓	-2.700e-06	1.906e-06	2.0068	1	0.156590
Test				χ^2	*df*	*p*
Overall model evaluation						
Likelihood ratio test				110.07	5	**< 2.2e-16**

Note. R programming: [package:" rcompanion","car","lmtest"]
McFadden R^2 = 0.147706
Model Null AIC = 747.22
Model Final AIC = 647.2

Appendix A4.

Final Logistic regression model of northern lapwing						
Northern lapwing~ Predictor	**Direction of effect**	**β**	**SE β**	**Wald's χ^2**	***df***	***p***
Fully cultivated soil	↑	3.089e-06	1.025e-06	9.0869	1	**0.00257**
Built up area	↓	-1.681e-05	9.228e-06	3.3173	1	0.06855
Open land	↓	-1.115e-05	5.809e-06	3.6816	1	0.05502
Test				χ^2	*df*	*p*
Overall model evaluation						
Likelihood ratio test				25.861	3	**1.02e-05**

Note. R programming: [package:" rcompanion","car","lmtest"]
McFadden R^2 = 0.0998706
Model Null AIC = 260.95
Model Final AIC = 241.1

Appendix A5.

Final Logistic regression model of white wagtail						
White wagtail~ Predictor	**Direction of effect**	**β**	**SE β**	**Wald's χ^2**	***df***	***p***
Infrastructure	↓	-3.607e-05	1.326e-05	7.4022	1	**0.00651**
Forest	↓	-1.015e-06	5.583e-07	3.3058	1	0.06903
Pasture	↓	-8.589e-06	3.900e-06	4.8513	1	**0.02763**
Shannon index	↑	6.884e-01	4.397e-01	2.4513	1	0.11743
Freshwater	↑	1.821e-06	1.199e-06	2.3062	1	0.12886
Test				χ^2	*df*	*p*
Overall model evaluation						
Likelihood ratio test				21.204	5	**0.0007411**

Note. R programming: [package:" rcompanion","car","lmtest"]
McFadden R^2 = 0.0311750
Model Null AIC = 682.18
Model Final AIC = 671

Appendix A6.

Final Logistic regression model of eurasian skylark						
Eurasian skylark~ Predictor	**Direction of effect**	**β**	**SE β**	**Wald's χ^2**	***df***	***p***
Fully cultivated soil	↑	9.424e-06	1.398e-06	45.4495	1	**1.57e-11**
Fractal dimension index	↓	-9.983e+01	3.820e+01	6.8297	1	**0.008965**
Open land	↑	3.493e-06	2.156e-06	2.6251	1	0.105186
Test				**χ^2**	***df***	***p***
Overall model evaluation						
Likelihood ratio test				103.42	3	**< 2.2e-16**

Note. R programming: [package:" rcompanion","car","lmtest"]
McFadden R^2 = 0.350745
Model Null AIC = 296.85
Model Final AIC = 199.4

Appendix A7.

Final Logistic regression model of yellowhammer						
Yellowhammer~ Predictor	**Direction of effect**	**β**	**SE β**	**Wald's χ^2**	***df***	***p***
Fully cultivated soil	↑	1.124e-05	1.286e-06	76.368	1	**< 2e-16**
Fractal dimension index	↓	-7.561e+01	1.638e+01	21.295	1	**3.94e-06**
Infrastructure	↑	9.156e-05	2.355e-05	15.116	1	**0.000101**
Test				**χ^2**	***df***	***p***
Overall model evaluation						
Likelihood ratio test				229.53	3	**< 2.2e-16**

Note. R programming: [package:" rcompanion","car","lmtest"]
McFadden R^2 = 0.341178
Model Null AIC = 674.75
Model Final AIC = 451.2

Appendix A8.

Final Logistic regression model of whinchat						
Whinchat~ Predictor	**Direction of effect**	**β**	**SE β**	**Wald's χ^2**	***df***	***p***
Pasture	↑	1.056e-05	3.739e-06	7.9786	1	**0.004733**
Infrastructure	↓	-3.974e-05	1.593e-05	6.2216	1	**0.012620**
Fully cultivated soil	↑	2.358e-06	6.777e-07	12.1004	1	**0.000504**
Surface cultivated soil	↑	2.944e-05	1.614e-05	3.3272	1	0.068143
Shannon index	↑	8.775e-01	4.702e-01	3.4828	1	0.062008
Open land	↑	1.381e-06	7.714e-07	3.2043	1	0.073445
Test				**χ^2**	***df***	***p***
Overall model evaluation						
Likelihood ratio test				41.384	6	**2.433e-07**

Note. R programming: [package:" rcompanion","car","lmtest"]
McFadden R^2 = 0.0623930
Model Null AIC = 665.28
Model Final AIC = 635.9