

# Negative population trend of the Reed Bunting *Emberiza schoeniclus* in an important wetland in southeastern Norway

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## Abstract

The Reed Bunting *Emberiza schoeniclus* is considered the most common of the five species of *Emberiza* buntings breeding in Norway and is currently listed as a species of Least Concern. However, a complete census of territorial male Reed Buntings conducted in the Nordre Øyeren nature reserve in 2019 showed a decline in the local breeding population of 67% compared to the last counts completed 40 years ago. Censuses were conducted in 1976 and 1981 in the same area and with the same methods. Analyses of citizen science data from the Norwegian Biodiversity Information Centre also show a similar decreasing trend (-66%) for the Reed Bunting breeding population in Nordre Øyeren from 1976 to 2019. Moreover, image analyses of the census maps show that the number and density of Reed Bunting territories have decreased locally and probably has done so since early 1980s. Interestingly, a subset of localities within the survey area still have the same territory densities as the 1976 and 1981 censuses. We suggest that changes in habitat, vegetation and farming practice, especially grazing, could be the underlying cause of variation in bird numbers. Our results indicate a negative population trend for the Reed Bunting locally, a tendency that is also observed regionally in the Nordic countries. Taken together, local and regional declines raise concerns about the conservation status of the Reed Bunting in Norway.

## INTRODUCTION

Population declines observed among several species of *Emberiza* buntings are cause for conservation concern. Four of the five *Emberiza* species breeding in Norway are currently red-listed as Vulnerable to Critically Endangered (Artsdatabanken 2021). The Reed Bunting *Emberiza schoeniclus* is considered the most common of the five species, and is currently categorized as a species of Least Concern on the Norwegian Red List, based on the population trends over the last 10 years (three generations; Artsdatabanken 2021). Data from the standardized, nationwide, area-representative terrestrial breeding bird surveys; the Norwegian Breeding Bird Assessment (NHT; Norsk hekkfugltaksering; Husby & Stueflotten 2009), and the Terrestrial Nature Monitoring (TOV; Terrestrisk naturovervåking; Kålås et al. 2014), nevertheless show a decrease of close to 40% in the 24-year period of 1996–

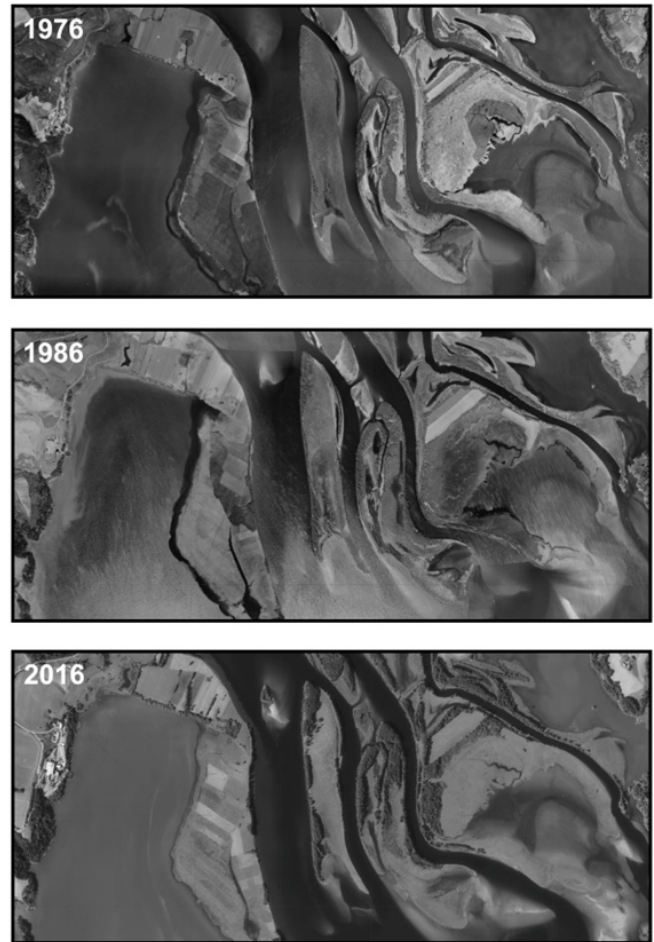
2019. The decline is consistent with population trends in the other Nordic countries in the last 10 years, which show decreases from 15 to 50% (Väisänen et al. 2018, Hyvärinen et al. 2019, Moeslund et al. 2019, Moshøj et al. 2019, Green et al. 2020). The Norwegian breeding population of Reed Bunting, which is estimated to 400,000–1,000,000 individuals, was considered to be in decline (Shimmings & Øien 2015), and was previously placed in the category Near Threatened on the 2015 national Red List (Henriksen & Hilmo 2015). In 2021, however, the Reed Bunting was down-listed and is now categorized as Least Concern (Artsdatabanken 2021).

In Norway, the Reed Bunting is widely distributed in suitable habitats throughout most of the country. In the mountains and forest areas at higher elevations, the Reed Bunting often nests in willow thickets along rivers, freshwaters and in marshy areas (Svensson et al. 2010). These habitats are where one finds the largest populations in Norway and the Nordic countries (Reitan

1994). Further south in the distribution area, Reed Buntings are associated with reedbeds and nutrient-rich wetlands (Cramp & Perrins 1994, Gregory & Baillie 1998, Surmacki 2001). The species' preferences for these habitats are presumably the reason why Reed Bunting populations in southern parts of Sweden and Norway grew in the post-war years. In this period, increased agricultural runoff led to eutrophication of wetlands which resulted in reed bed expansion (Fremstad & Moen 2001, Lundekvam et al. 2003), while reeds and shrubs were also allowed to regrow on abandoned pastures and hayfields (Haftorn 1958, Larsson 1969).

The Nordre Øyeren delta is created by the confluence of the Leira, Nitelva and Glomma rivers. The delta covers an area of 62.6 km<sup>2</sup> and was designated a nature reserve in 1975 and then as a Ramsar site in 1985. The site is northern Europe's largest inland delta and it is an important wetland in eastern Norway (Akershus fylkeskommune 2002). Historically, Lake Øyeren was regulated in 1862 to secure water levels for log driving and to make the farming more predictable. Later, a hydroelectric power dam was built in the 1920s. Before regulation, the variation in water levels throughout the year was up to 10 meters. After regulation was implemented, the minimum water level was raised by four meters and the seasonal variation was considerably reduced. Regulation has caused sediments to accumulate further north in the lake, and thus the current land area of the delta has more than doubled compared to historical maps from before regulation was implemented (Horgen 1991, Bogen et al. 2002). For ground-nesting birds, such as Reed Bunting, reduced variation in water levels has probably been advantageous, even if floods during May-June still destroy many nests in some years.

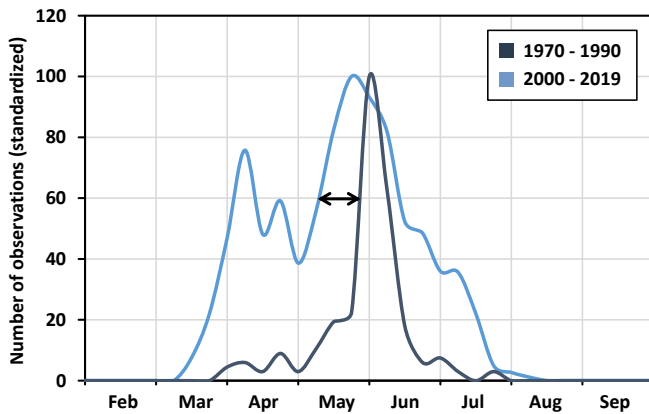
The natural vegetation in most of the Nordre Øyeren nature reserve is deciduous or mixed forest. Thickets of various species of willow (*Salix* spp.) grow along the banks and on newly formed islands. Much of the land area in the reserve has been used for agriculture for many years, and the traditional land use was a combination of mowing and grazing of natural meadows and grasslands. In the post-war years, however, the traditional mowing ceased, and today only a few small areas of natural meadow are mowed (Regionkontor Landbruk 2013). From the 1960s onwards, most of the higher-lying meadow areas were cultivated and used mainly for grain production. Mowing ceased on the lower-lying areas, and in addition, only a few smaller patches were grazed until 1990s. The changes in land use allowed the meadows to regrow with shrubs (Figure 1). In an attempt to prevent shrub encroachment, most of the meadow areas were burned annually until 2006. However, because burning is not considered an acceptable management method in wetlands (Blindheim & Olsen 2008) and because grazing increased as a preferred farming system in the reserve, the practice of prescribed burns was discontinued in 2007. Larger areas of the reserve are now grazed, more



**Figure 1.** A changing landscape. Changes in agricultural production modes, discontinuation of mowing and reduced grazing up until the 1990s, have led meadows to regrow with shrubs and trees. From 1976 to 2019, several shrub and forest areas have appeared on the islands in the central parts of the delta. From left to right: Årnestangen, Storsand, Bukkesand, Rossholmen and Jørholmen. See Figure 3 for a complete delta map with island names.

than has been done in 50 years.

The Reed Bunting was a common species in the reserve when Nordre Øyeren Bird Observatory began its field registrations in the late 1960s, and regular observations of large numbers of birds are well documented ([www.artsobservasjoner.no](http://www.artsobservasjoner.no), NØF 1977). In the period 1973-1975, the Reed Bunting was by far the most common among the ringed species of birds at Nordre Øyeren, with a total of 1,728 of all 5,432 ringed individuals (32%), followed by the Willow Warbler *Phylloscopus trochilus* with 408 individuals (8%, NØF 1977). Most of the Reed Buntings were likely ringed while migrating through Nordre Øyeren, but two breeding population censuses in the same area and period (1976 and 1981) reported good numbers of singing males (Syvertsen 1985). At the time, the Reed Bunting was considered a characteristic species for the flat delta landscape during summertime. In Nordre Øyeren, the species is found in a variety of vegetation types ranging from partly bushy meadows and field edges to areas with quite dense shrub. Forests and



**Figure 2.** Relative numbers of singing Reed Bunting males observed per week in Oslo, Akershus and Østfold counties between 1970 and 1990 ( $n = 186$ ), and from 2000 to 2019 ( $n = 2,414$ ). The double-ended, black arrow indicates the 2-3 week shift in the most intense song activity from 1970-1990 to 2000-2019.

completely open meadows are avoided because male Reed Buntings use bushes as singing and lookout posts when marking their territories (Syvertsen 1985).

In this paper, we present data from a new complete census of territorial male Reed Buntings in 2019, and compare our results with the two earlier censuses conducted ca. 40 years ago in 1976 and 1981. In addition, using 45 years of bird observation data from Nordre Øyeren, we analyse long-term changes in the breeding and migratory populations of the Reed Bunting, looking for evidence of when the decline of the species started in the nature reserve. Using map data from the three complete censuses, we also examine how the patterns of territory density have changed during the years. The cause of the decline in the breeding population is not known. Nevertheless, we discuss some of the changes that have occurred in the area during the last 45 years and evaluate what factors may have influenced on habitat conditions. For this question, it is noteworthy that there seem to be a subset of localities where the Reed Bunting population has not declined.

## METHODS

### *Reed Bunting Census*

When the Reed Bunting was placed in category Near Threatened on the Norwegian Red List in 2015 (Henriksen & Hilmo 2015), Nordre Øyeren Bird Observatory decided to conduct a new census of Reed Buntings in the nature reserve. A complete census of all singing Reed Bunting males was conducted on all land areas in Nordre Øyeren nature reserve in the spring of 2019. The census technique was the same as for two earlier surveys of the same area, conducted in 1976 and 1981 (Syvertsen 1985). Briefly, singing Reed Buntings were recorded along predefined routes within predefined survey units of islands, landspits, and shores. The survey routes were drawn based on natural borders, habitats and agricultural areas. The

aim was that the observers should never be more than a maximum of 50 m from any point in the terrain. One exception was thick forest, which was not surveyed as it is considered an unsuitable breeding habitat. Field personnel recorded all singing males, as well as observations of breeding Reed Buntings in courtship, mating, or nest building activities, and plotted the observations directly on field maps while surveying. A total of 25 routes were defined, dividing the total survey area into 13 sub-localities representing single or naturally interconnected islands, peninsulas, landspits and shorelines within the survey area, enclosed by water and roads. Standardized routes were used to keep the same sampling effort as the censuses in 1976 and 1981 (Syvertsen 1985), making the data directly comparable, and to be able to detect any changes that might be specific for some of the sub-localities. Importantly, the sub-localities often represented an area where one or a limited number of farmers employed one particular farming system.

The censuses were completed between 11 May and 17 May 2019 in the early morning (02:30–06:30). The timing of censuses was somewhat earlier in the year in terms of date but somewhat later in the morning than when censuses were carried out in 1976 (26 May–8 June; 01:15–04:00) and 1981 (31 May–10 June; 02:20–04:20). The dates were shifted to adjust for long-term changes in the timing of spring migration, arrival of the local population, and most intense song activity of Reed Buntings that occurred 2–3 weeks earlier in 2019 as compared to the 1970s/80s (Figure 2; see also paragraph Reed Bunting song season analysis) (Jonzén et al. 2006, Thorup et al. 2007, Lehtikoinen et al. 2019). Consequently, the field recordings began somewhat later in the morning, to adjust for a later sunrise in the middle of May (2019), than at the end of May (1976 and 1981). The field personnel carefully assessed the survey routes, either alone or in teams of two observers. All routes were completed in one given morning. For three of the sub-localities, the routes were surveyed twice on two different dates in 2019 to check whether weather conditions and time of day affected the results. The results from the three replicate surveys were 3 vs 4, 8 vs 7, and 0 vs 0 singing males. Thus, we concluded that the data from the surveys were robust and reproducible.

### *Reed Bunting song season analysis*

To determine how the seasonality of Reed Bunting song activity might have changed from 1976 to 2019, all observations of singing Reed Buntings in Oslo, Akershus and Østfold counties between 1970 and 1990 ( $n = 186$ ), and from 2000 to 2019 ( $n = 2,414$ ) were downloaded from [www.artsobservasjoner.no](http://www.artsobservasjoner.no) (the Norwegian Biodiversity Information Centre). The activity „Display/Song” (or “Sang/spill i hekketid og passende hekkebiotop”) was used as a search criterion. The data were binned by week, before the two data series (1970–1990 and 2000–2019) were standardized by setting the maximum value for both to 100.

### **Image analysis of territory maps**

To test for changes in the spatial distribution of the breeding population of Reed Buntings, we digitally analysed the territory maps for each of the years 1976, 1981 and 2019. The density of territories within the survey area was analysed using the image-processing program Fiji (Schindelin et al. 2012) for calculations of the distance to the nearest singing male (Nearest Neighbour Distance; NND) and the number of singing males within a radius of 200 m (Number of Neighbours Within Distance; NNWD). For each of the years 1976, 1981 and 2019, locations on the digital maps corresponding to singing males were identified and mapped (X-Y coordinates) using the „analyse particles” (NND) or „3D Objects Counter” (NNWD) functions in Fiji. Thereafter, distances to all other singing males were measured as the linear distance between territory centres and ranked using the „Nearest Neighbour Distances” plugin written by Yuxiong Mao to find the NND. To determine the NNWD singing males within a 200 m radius from each territorial male were counted, using the „NumNeighbWithinDist” macro written by Ofra Golani for the 3D Suite (Ollion et al. 2013). The NNWD set distance was chosen based on the prior NND analysis, showing an upper aggregated cluster of around 200 m, mainly from the 2019-data, allowing the NNWD analysis to capture the differences in territorial spacing among the three censuses.

### **Mapping of Reed Bunting territories on Fautøya 2010.**

As part of a complete territory mapping of breeding birds on the island Fautøya in Nordre Øyeren in 2010 (NØF 2010), all Reed Bunting territories was mapped using simplified territory mapping, as described in the Bird Census Techniques handbook (Bibby et al. 2000). Fautøya was visited six times between 15 May and 30 June, and the field personnel surveyed the entire island following a route that was drawn based on natural borders, habitats and agricultural areas. The aim was that the observers should never be more than a maximum of 50 m from any point in the terrain. The surveys were conducted in the morning (05:30–10:00). All types of observations were recorded and mapped, including information on species, sex, age, numbers, and bird activity (singing, feeding, roosting, nest building, or warning). Last, all survey maps were projected on top of each other and territories assigned following five simple rules: 1) a given species must have been observed more than once within the survey area to be assigned a territory, 2) a territory must include at least two independent observations on at least two separate dates, 3) the territory boundaries must fall as far as possible between simultaneous observations, 4) a single observation must be added to the nearest territory as long as this does not increase the territory size disproportionately much, and 5) strong evidence of breeding activity, such as observations of nests in use or feeding of young may allow deviations from rules 1 and 2.

### **Trend analyses of migrating and breeding Reed Buntings using citizen science data**

Observational data were downloaded from [www.artsobservasjoner.no](http://www.artsobservasjoner.no) for the three most visited landspots in the Nordre Øyeren nature reserve: Årnestangen, Jørholmen and Tuentangen. The annual maximum count, defined as the average of the five highest daily numbers during spring migration (1 March–14 May) and breeding season (15 May–30 June), was calculated for each locality and for each year between 1976 and 2019. To be included in the analyses, the observations had to fulfil the following criteria. All checklists with Reed Bunting observations were included if the Reed Buntings were counted (numerical or “not found” = 0). Checklists where Reed Bunting were only recorded as “found” were excluded. All checklists without Reed Bunting observations were included and the tally set to zero, if the total number of species recorded on the same checklist were  $\geq 15$ . The latter rule was based on background data on relative abundance, where every 13–14th bird observed in the same area during the same period was a Reed Bunting (26,581 of 356,019), and overall every 17–18th observation was a Reed Bunting observation (1,893 of 33,571). Importantly, the filtering criteria also allowed us to include the data from our three censuses in the calculations of the annual maxima. Years with no Reed Bunting observations based on the filtering above, were excluded from the trend analyses (Årnestangen: zero; Jørholmen: 11; Tuentangen: 17).

### **Statistics**

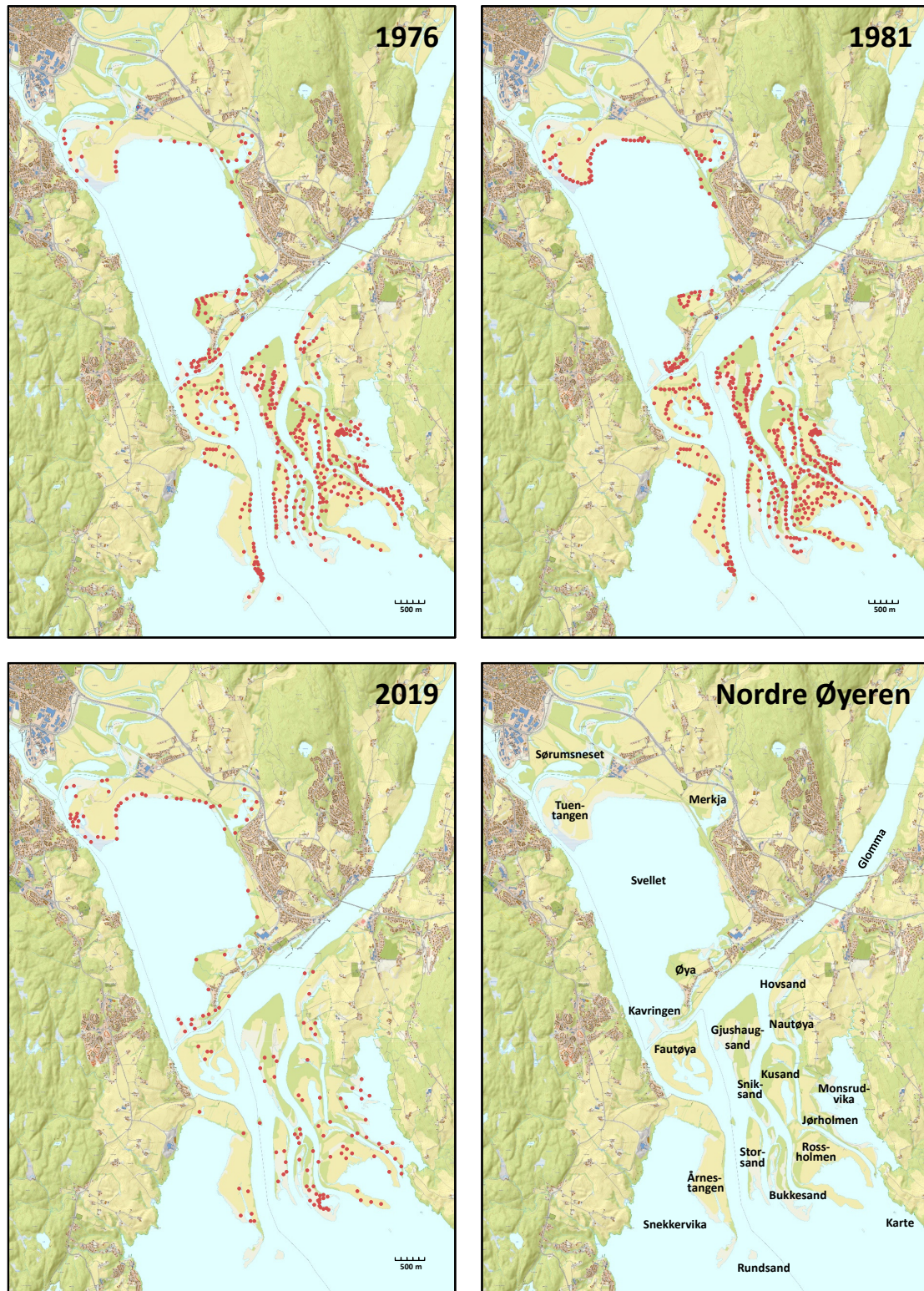
To identify trends in the Reed Bunting spring migration numbers and breeding season numbers, we analysed the maximum counts described above using a generalized linear mixed model (GLMM) with Poisson probability distribution. The analysis was conducted in IBM SPSS Statistics v. 29.0 (IBM, Armonk, NY), where the number of annual visits to each sub-locality was included as a covariate, and sub-locality included as a random factor, circumventing the need for normalizing the data from the three sub-localities.

Statistical comparisons of the differences in spatial distribution among the 1976, 1981 and 2019 breeding populations (NND and NNWD) was calculated using one-way analysis of variance (ANOVA) in GraphPad Prism v. 9.3.1 (GraphPad Software, San Diego, CA), with posthoc pairwise comparisons of means (Tukey HSD test).

The changes from 1976 and 1981 to 2019 for the different sub-localities were calculated in GraphPad Prism using unpaired, two-tailed t-test. Statistical tests were considered significant at an  $\alpha$ -level  $< 0.05$ .

### **Data availability**

The historical orthophotos in Figure 1 were downloaded from <https://kart.finn.no>. The Reed Bunting data in Figure 2 can be downloaded from the Norwegian Biodiversity Information Centre (NBIC), using the following Search Stored Query (SSQ) ids: [6780129](#) for



**Figure 3.** Maps of locations of singing Reed Bunting males in Nordre Øyeren nature reserve. Data from the 1976, 1981 and 2019 censuses are shown with a background map with sub-locality names. Each red dot represents a singing male.

the years 1970-1990 and [6780148](#) for the years 2000-2019. The data from the Reed Bunting censuses in 1976, 1981 and 2019, used in Figures 3, 4, 7 and 8, can be downloaded from NBIC using the SSQ [1382101](#). The Reed Bunting data from Nordre Øyeren's three most visited landspits, used in Figure 5, can be downloaded from NBIC using the SSQ [8161665](#) (Årnestangen), SSQ

[8161666](#) (Jørholmen), and SSQ [8161667](#) (Tuentangen), whereas the data from the breeding bird survey in 2010, used in Figure 6, can be downloaded in the same site using SSQ [6781360](#). All data not mentioned in this section are available upon request.

**Table 1.** Test statistics of the Generalized Linear Mixed Model (GLMM). Factors considered statistically significant ( $P < 0.05$ ) are outlined.

Model factors	Estimate	SE	t (Z)	P
<i>Target: Annual maximum (Migration)</i>				
Year (fixed)	0.000	0.0059	- 0.017	0.986
No. of visits (fixed)	0.002	0.0033	0.583	0.561
Sub-locality (random)	0.831	0.835	0.996 <sup>a</sup>	0.319
<i>Target: Annual maximum (Breeding)</i>				
Year (fixed)	- 0.024	0.0073	- 30.33	<b>0.001</b>
No. of visits (fixed)	0.023	0.0059	30.84	<b>&lt; 0.001</b>
Sub-locality (random)	0.277	0.284	0.975 <sup>a</sup>	0.330

Notes: Estimate = estimated coefficients from the GLMM with Poisson probability distribution. SE = approximate standard error. t (Z) = ratios between the estimates and their standard errors (fixed effects) or <sup>a</sup>Wald Z test statistics (random effects).

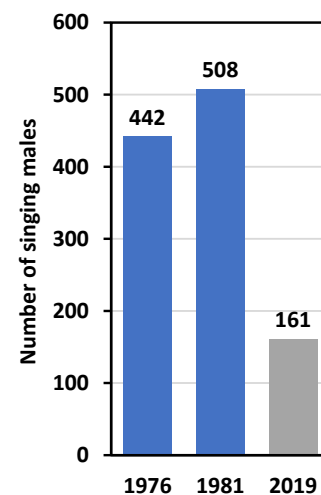
## RESULTS

A complete census of singing Reed Bunting males was conducted on all land areas in Nordre Øyeren nature reserve in the spring of 2019 and compared to two earlier surveys of the same area from 1976 and 1981 (Syvertsen 1985). We used the same survey methods with one exception: the survey dates were shifted two weeks from late to early May, to adjust for the earlier arrival of the local population and to coincide with the most intense song activity of Reed Buntings, which occurred 2–3 weeks earlier in 2000–2019 as compared to 1970–1990 (Figure 2). The 2019 survey showed a dramatic decline of Reed Bunting territories in Nordre Øyeren. When all recorded observations were mapped (Figure 3) and counted, the results were 161 singing males with a decline of 67% from 442–508 males in 1976/81 (Figure 4). Similar patterns of decline are also seen in other types of observation data from Nordre Øyeren. When analysing citizen science data from the Norwegian Biodiversity Information Centre using a generalized linear mixed model, we identified a significant reduction in the annual maximum of Reed Buntings observed during breeding season (15 May–30 June) at the three most visited landspits in Nordre Øyeren nature reserve; Årnestangen, Jørholmen and Tuentangen (Figure 5 lower panel; Table 1). Using the estimated Poisson coefficient (Estimate; Table 1) to calculate the rate ratio for the 44-year change in annual maximum (Hilbe 2011), indicated a 66% drop in the number of Reed Buntings observed between 1976 and 2019 ( $[e^{\text{estimate}}]^{(2019-1976)} = 0.34$ ). The decline could not be explained by the number of visits to these sites, as the number of visits per year increased from 1976 to 2019 and showed a positive correlation with the annual maximum (Table 1). No significant change was observed during spring migration (1 March–14 May) in the same dataset (Table 1; Figure 5 upper panel). The decline in the breeding population was also supported by data from the island Fautøya (Figure 6), where Nordre Øyeren Bird Observatory carried out an extensive territory mapping of breeding birds in 2010

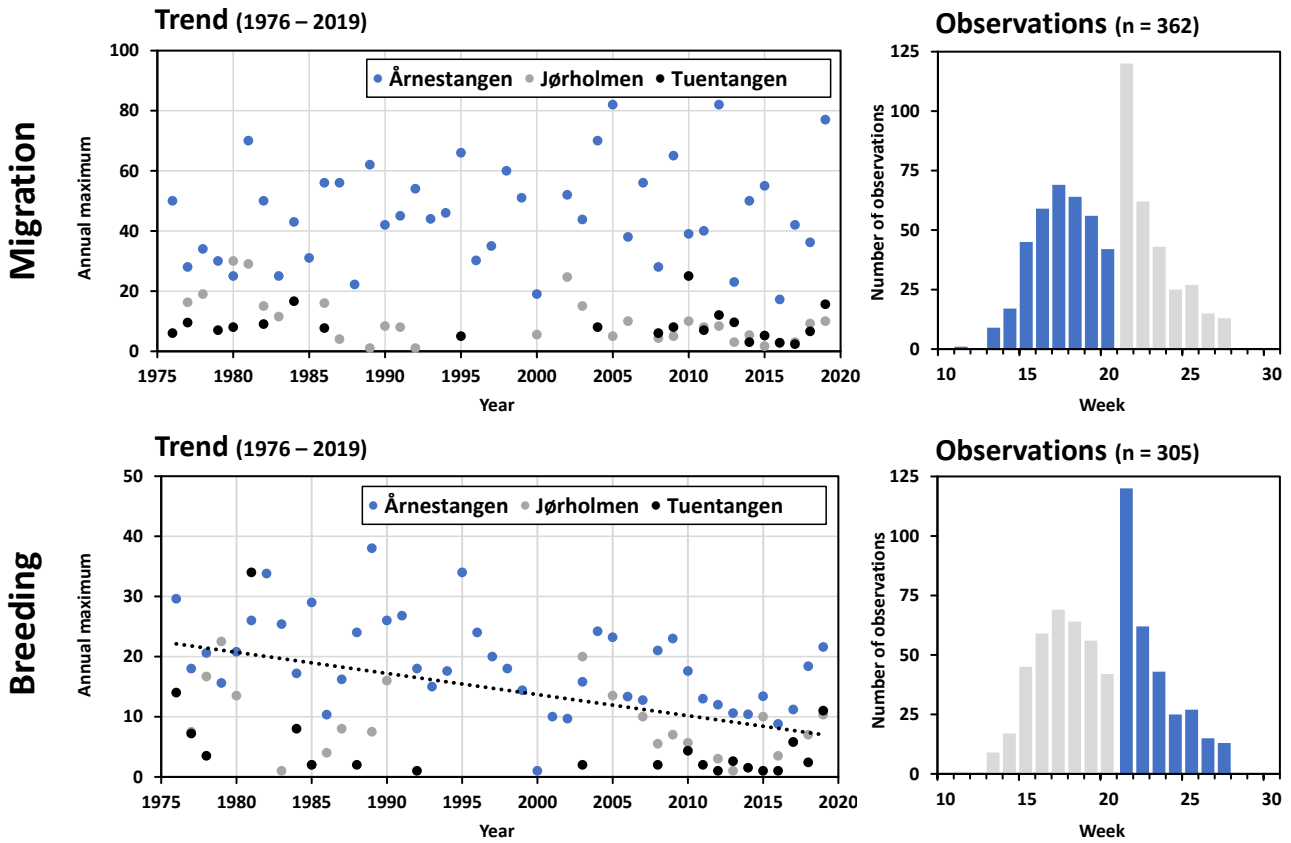
(NØF 2010). Together, the data suggest that the decline in the Reed Bunting breeding population in Nordre Øyeren has progressed steadily since 1976/81.

In parallel with the overall decline in the breeding Reed Bunting population, the spatial distribution of territories also changed from 1976/81 to 2019. The average distance to the nearest singing male (NND; Nearest Neighbour Distance) showed an increase from 90.1 m and 76.9 m in 1976 and 1981, respectively, to 140 m in 2019 (Figure 7a). At the same time, the number of singing males within a radius of 200 m (NNWD; Number of Neighbours Within Distance) fell from 6.8 and 8.3 males in 1976 and 1981, to 3.8 males in 2019 (Figure 7b). Since both NND and NNWD decreased from 1976/81 to 2019, we conclude that both the number and density of Reed Bunting territories have decreased in a large part of the investigated area.

While we observed a significant decline in the Reed Bunting population in Nordre Øyeren nature reserve, it appears that certain sub-areas have retained a territory density similar to what was observed in 1976/81 (Figure 7b; upper cluster 2019). When examining the 2019 data based on sub-locality; single or naturally interconnected



**Figure 4.** Numbers of singing Reed Bunting males in the surveyed area for the years 1976, 1981 and 2019.



**Figure 5.** Changes in Reed Bunting numbers, as represented by maximum counts per year observed during spring migration (1 March–14 May) and the breeding season (15 May–30 June) at the three most visited landspits in the Nordre Øyeren nature reserve. Left panels: The data were analysed using a generalized linear mixed model (GLMM) with Poisson probability distribution and sub-locality as a random factor. See Table 1 for test statistics. Regression line for the breeding population, for which the changes was found to be statistically significant, is included for visual purposes [ $f(x) = -0.35x + 716$ ]. Right panels: plots showing the distribution of observations analysed by GLMM, binned by weeks (blue bars). Migration  $n = 362$ ; breeding  $n = 305$ .

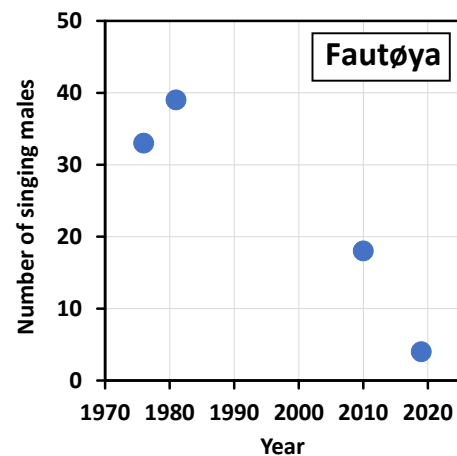
islands, peninsulas, landspits and shorelines within the survey area, it was evident that the numbers of singing males have decreased significantly from 1976/81, except for three sub-localities: A) Tuentangen, B) Merkja and Svellet, and C) Bukkesand and Midsand (Figure 8). Interestingly, by ranking the sub-localities by percent change and colour-coding the bars based on grazing status, a pattern appears. The grazed areas seem to cluster at the lower end of the scale with the most dramatic declines, whereas the opposite is true for ungrazed localities.

**DISCUSSION**

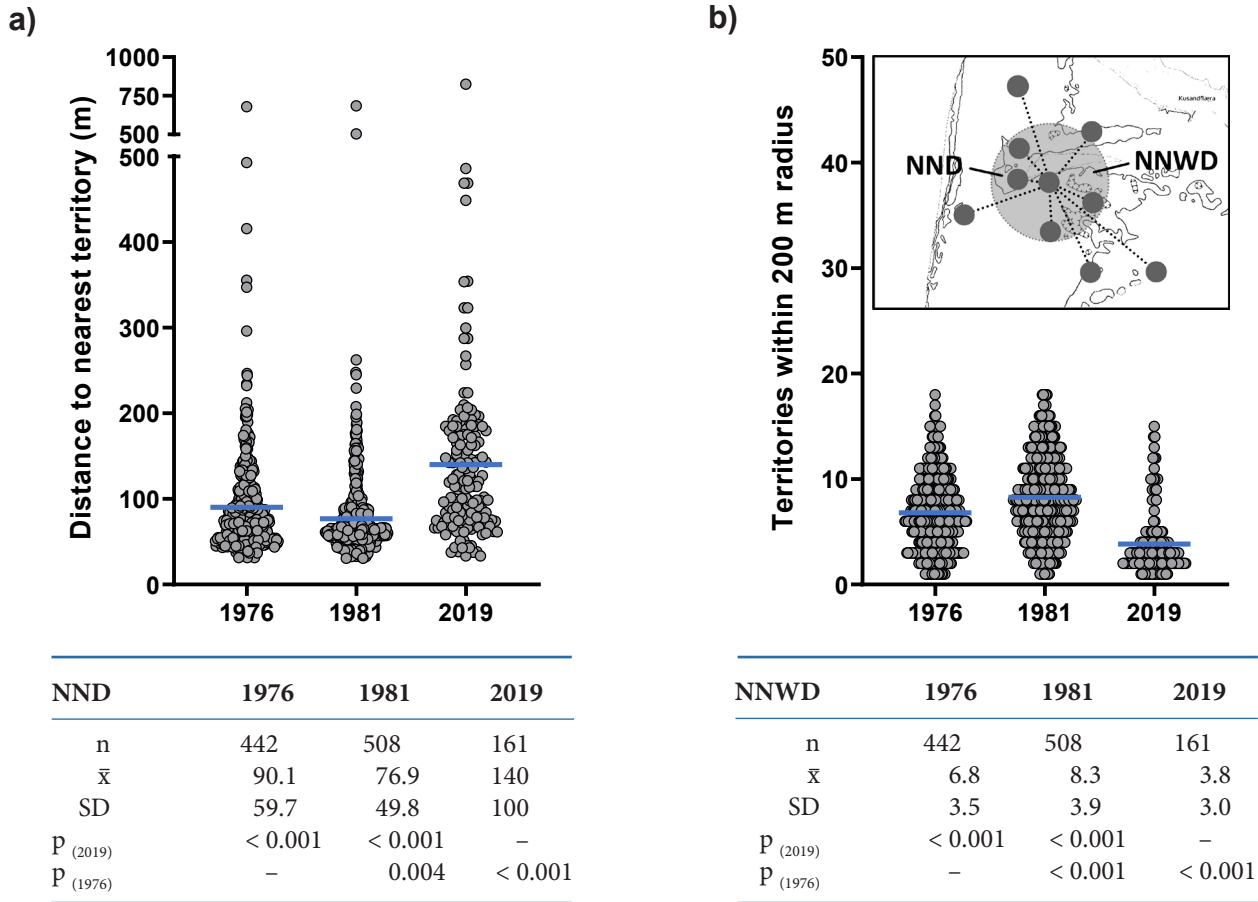
The Reed Bunting census in 2019 revealed a dramatic decline in the number of territories in Nordre Øyeren nature reserve since 1976/81. The change in territorial males was mirrored by a similar decline in observations of Reed Bunting during the breeding season. Noticeably, our data showed a significant difference between different sub-localities, where 10 of 13 sites had experienced a reduction in numbers and densities of singing males since 1976/81, while the remaining three seemed to be unaffected by the decline. Differences in

habitat suitability, vegetative structure and farming practices among the sub-localities might point to the underlying cause of variation.

Since the first censuses of Reed Bunting were conducted in 1976 and 1981, several areas with suitable Reed Bunting habitat have grown back with trees (Figure 1). Afforestation appears to be the case at



**Figure 6.** Changes in the number of singing Reed Bunting males at Fautøya from 1976 to 2019. Territory mapping data from 2010 were included (NØF 2010).



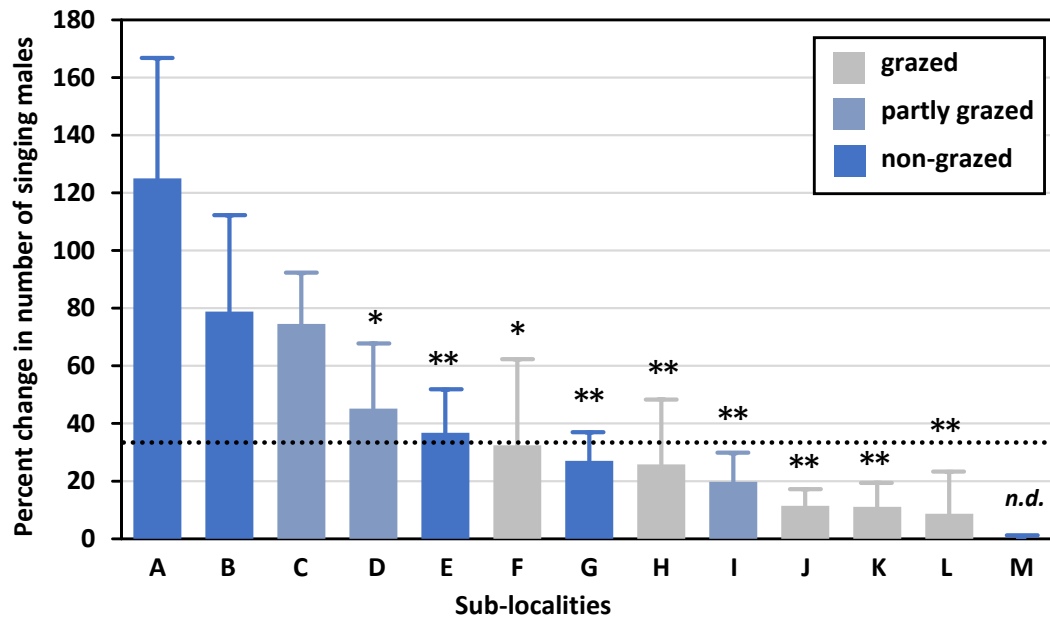
**Figure 7.** Spacing and density of Reed Bunting territories within the survey area. The figure shows a) distance to the nearest singing male (Nearest Neighbour Distance; NND) with statistics), and b) number of singing males within a 200 m radius distance (Number of Neighbours Within Distance; NNWD) with statistics). The map section in b) shows the NND and NNWD (light grey circle) for a specific Reed Bunting territory (centred point). n = number (population), blue lines (–) and  $\bar{x}$  = mean, SD = standard deviation, p(year) = significance level between column year and year in brackets.  $p < 0.05$  was considered statistically significant, as determined by one-way ANOVA with Tukey's multiple comparisons test.

Kusand, where singing Reed Buntings were recorded in 1976, but where today there is tall forest (Figures 3 and 8, J) that is an unsuitable habitat for the species. Both tall forests and completely open meadows are avoided because male Reed Buntings use bushes as singing and lookout posts when marking their territories. Still, a positive association between Reed Bunting density and proximity to forest (250 m) has been reported earlier (Žmihorski et al. 2016).

At the other end of landscape changes, grazing has increased greatly as a viable farming system in the reserve over the last 30 years, and at the same time been recognized as a method for wetland management, as grazing helps to constrain expansion of shrub vegetation. As such, grazing could theoretically help to retain or improve habitat quality for Reed Buntings. However, grazing might be counterproductive with respect to breeding success for ground-nesting birds, as certain species, like cattle, are known to trample and destroy bird nests (Pavel 2004, Sabatier et al. 2015, Brambilla et al. 2020). The effect of grazing on Reed Bunting densities is somewhat inconclusive, possible

due to different types of grazing management with variation in animal species, time of grazing, and type of grassland (wet or dry) (Vera et al. 2011, Kerekes & Végvári 2016, Žmihorski et al. 2016). Certain types of livestock grazing might also be unfavourable if they reduce the tall grass and sedge where Reed Buntings nest. Interestingly, the 2019 census showed a strong decrease in Reed Bunting territories on the islands where grazing animals are allowed to roam freely, such as Rossholmen (H), Kusand (J), Fautøya (K), and Sniksand and Gjushaugsand (L; Figure 8). The areas without declines in Reed Bunting numbers are primarily ungrazed, with the exception of certain parts of Merkja (Figure 8, C), and they have little to no forest. It looks as if these areas have reached a stage of succession from open meadow to forest that is favourable for the Reed Bunting (Figure 9). However, to establish a causative relationship between succession stage and Reed Bunting territories, a more extensive vegetation survey is warranted. Importantly, the livestock herds were released on the islands and pastures after the Reed Bunting censuses in 1976, 1981, and 2019.





**Figure 8.** Percent change ( $\pm$  SE) in numbers of singing Reed Bunting males from 1976/80 to 2019 at 13 different sub-localities in the survey area (A-M). The grey columns represent sub-localities that are grazed, slate grey represent sub-localities that are partly grazed, while the blue columns represent areas that are ungrazed. The dotted horizontal line shows the average for 2019 for the entire survey area/all sub-localities. \* $p < 0.05$ , \*\* $p < 0.01$  as determined by unpaired, two-tailed t-test. n.d. = not determined. Sub-localities: A) Tuentangen, B) Bukkesand and Midsand, C) Merkja and Svellet, D) Hovsand and Nautøya, E) Øya and Kavringen, F) Storsand, G) Jørholmen and Monsrudvika, H) Rossholmen, I) Årnestangen, Staka- and Rundsand, J) Kusand, K) Fautøya, L) Sniksand and Gjushaugsand, and M) Karte.

Consequently, any correlation between grazing and the Reed Bunting density in Nordre Øyeren, detected by our survey, should be attributed to long-term, rather than immediate, effects of grazing practices.

Another possible explanation for the variation in declines among sub-localities could be the height above normal summer water level (HRV; 4.80 m). Low-lying areas are more prone to flooding and thus regrow more slowly than high-lying areas. With respect to flood impacts, we lack vegetation data that might support a causative conclusion. Importantly, we observed low-lying areas with a dramatic decline in the density of Reed Bunting (e.g., Årnestangen; Figures 3 and 8, I) but also low-lying areas where the number of singing Reed Bunting males remained high (e.g., southern part of Bukkesand; Figures 3 and 8, B). As a single factor, water levels cannot explain variation in bunting numbers, but could still be a contributing factor together with the conditions mentioned above. An obvious task for future surveys will be to clarify any associations between habitat types, vegetative succession, farming systems, water level management, availability of insects and other food resources, and the territory density of Reed Buntings.

Regardless of what might have caused the observed variation in territory density of Reed Buntings among the various sub-localities in Nordre Øyeren (Figure 8), it would be beneficial to determine whether the changes stem from a local decline only, or whether the reduction in the number of singing males may be caused by national or regional trends. Locally, altered natural

qualities might play an important role: Increased grazing, reduced shrub vegetation and tall grass on low-lying areas, accompanied by forest regrowth on high-lying areas, could be part of the explanation. However, the observed drop in local numbers of Reed Bunting territories, due to new farming systems or stages of succession, does not rule out a corresponding, or even contributing, decline in other parts of Norway. What is certain, is that the decline in Nordre Øyeren has left this important wetland with few patches where more than four singing males are found within a 200 m radius (Figure 7b). The areas with territory densities of  $> 4$  males/12.5 HA are reduced to small clusters with a relatively limited distribution (Figure 3).

By studying old aerial photographs, it is not obvious that the habitat changes in Nordre Øyeren during the last 50 years could explain all of the observed decline in the Reed Bunting breeding population (Figures 1, 3, and 4). Still, the 67% reduction seen in Nordre Øyeren is much higher than what has been reported for the species at a national level by the Norwegian Biodiversity Information Centre (Artsdatabanken 2021). As a result of the revision of the Norwegian Red List for Species in 2020–21, the Reed Bunting was down-listed from the category Near Threatened and is now categorized as Least Concern (Artsdatabanken 2021). The reclassification occurred despite the fact that spring and autumn migration data from Jomfruland and Lista Bird Observatories show a general, aggregate decrease of 49.5% in Reed Bunting numbers in the period 2010–2019 (Røer 2020), even



**Figure 9.** Sunrise over Tuentangen in Nordre Øyeren nature reserve. The tufted grassland or “tuemark”, scattered with willow shrub and reeds, has given Tuentangen its name, and has provided the Reed Bunting with excellent breeding grounds during all of the years NØF has conducted field registrations in the nature reserve. Photo: Trude Starholm

if the individual season-site series are not statistically significant. The Red List bird expert group wrote in their assessment that: „...a limitation with regard to such (migration) data, however, is that it is not possible to determine what proportion of the birds counted are birds that breed in Norway” (Artsdatabanken 2021). The statement is indeed true, but when combined with the data presented in this article, there are still reasons to be concerned about the conservation status of the Reed Bunting. Hence, our findings emphasise a need for reference areas for monitoring of red-listed species, including those species classified as Near Threatened or Vulnerable. Reference areas should be censused regularly using standardized methodology and the Nordre Øyeren nature reserve would be a suitable reference area for the Reed Bunting.

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