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Non-random sampling along rural-urban gradients may reduce reliability of multispecies farmland bird indicators and their trends

Running title: Sampling design effects on MSI trends

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The continued global biodiversity crisis necessitates the continuation and development of new well-designed monitoring strategies and action plans with special focus on underrepresented countries and regions. However, limited resources in terms of budget and availability of qualified field personnel can restrict the geographic coverage of monitoring efforts. Focusing monitoring efforts on a representative subset of species and locations can improve cost-efficiency. Optimal performance of multi-species indicators derived from such an approach requires objective methods for species selection and a sampling design that reduces inherent sampling bias caused by regional differences in habitat availability or accessibility. To explore the performance of a multi-species indicator across different regions within a nation, we develop a multi-species indicator (MSI) for farmland birds in Norway using objective niche-based selection of species. We compare the performance of this indicator at national and regional-scales (Central and East regions) in Norway, and between urban and rural sites within regions. The 7-species indicator obtained from the species selection provided similar indicator values and trends for Norway and the Central and East regions, as well as for Rural sites within the combined Central+East region. All trends were defined as showing moderate decline from 2007-2016. Urban sites within the combined Central+East region provided trend estimates that showed stronger decline than rural areas in the same region during the time span. Our results emphasise the need to control for sampling bias when structuring monitoring programs such as a Breeding Bird Survey (BBS). This is especially important if limited resources restrict the geographic coverage of the monitoring scheme. We recommend that monitoring schemes follow a stratified random sampling design that represent both the availability of different land cover types and their distribution with regard proximity to highly populated areas. If that is not possible, statistically weighting data from different regions or landscapes is likely to be necessary.

The growth and expansion of human populations places increasing demands on nature's resources, putting a high strain on natural environments, and global biodiversity is declining more rapidly now than ever in human history (IPBES 2019), ultimately reducing human well-being (Scholes *et al.* 2008). Several international conventions aimed at reducing or halting global species loss (e.g. Convention on Biological Diversity, Convention on International Trade in Endangered Species of Wild Fauna and Flora, Convention on the Conservation of Migratory Species of Wild Animals) have been put in place. These encourage or require signatory states to implement strategies and action plans for monitoring national biodiversity and issue national assessments of biodiversity (Secretariat of the Convention on Biological Diversity 2017).

Indicator species (Andelman & Fagan 2000, Dale & Beyeler 2001) or, more recently, multi-species indicators (hereafter MSIs) such as the Living Planet Index (Loh *et al.* 2005, McRae *et al.* 2017) and European Farmland Bird Index (Gregory *et al.* 2005), are increasingly used both to monitor the state of important ecosystems and wider biodiversity health, and to measure the impact of actions designed to mitigate the effects of detrimental environmental change (Pellissier *et al.* 2013). Effective biodiversity indicators need to fulfil several scientific and policy related requirements, including being a) representative of the ecosystem (Norris & Harper 2004, Butchart *et al.* 2010), b) quantitative, using metrics which are responsive to change while simultaneously buffering against irregular, large natural fluctuations, and c) easy to update and interpret (Gregory *et al.* 2005).

Birds fulfil many of the requirements of biodiversity indicator species and have become widely used in this context during the last few decades (Gregory *et al.* 2005, Gregory & Strien 2010, Wotton *et al.* 2017, Hoffmann *et al.* 2018). Birds are relatively simple to observe and identify, and we understand their ecology better than any other taxonomic group (Tucker 1997). In addition, they respond rapidly to environmental change (Ortega-Álvarez & Lindig-Cisneros 2012), and their abundance is assumed to reflect the availability of insects and other food resources (Furness & Greenwood 1993). Due to their charismatic nature, bird species receive a high level of interest from the public, which makes it easier to start and maintain monitoring actions as large amounts of data can be collected at relatively low cost through the involvement of volunteer ornithologists (Butchart *et al.* 2010, Stephens *et al.* 2016). For example, population trends of common bird species across Europe (European Bird Census Council 2017) and most of North America (Sauer *et al.* 2017) are calculated from large-scale monitoring data collected predominantly by volunteer ornithologists. MSIs, generated from the composite trends of species with similar ecology, have become indicators of sustainable development, biodiversity health and a structural indicator in many European countries (Butchart *et al.* 2010, Gregory & Strien 2010, European Bird Census Council 2017) and in North America (Hudson *et al.* 2017). Changes in bird populations are used to indicate the state of marine (Montevecchi 1993), woodland (Gregory *et al.* 2007), farmland (Tucker 1997, Gregory *et al.* 2005), peatland (Fraixedas *et al.* 2017) and mountain environments (Lehikoinen *et al.* 2014, 2019), water quality (Ormerod & Tyler 1993), and the effects of pollution (Furness 1993) and climate change (Stephens *et al.* 2016).

A crucial step in ensuring that an indicator possesses all the required attributes to be effective is the selection of which species to include (Gregory *et al.* 2019). Species selection can affect indicator performance and projections about ecosystem state, with inappropriate indicators potentially providing misleading results (Lindenmayer *et al.* 2000). However, indicator species selection frequently relies on expert opinion (Gregory *et al.* 2005, Sætersdal *et al.* 2005, Husby & Kålås 2011), and clear details of any framework used to support this process are often not available (Hilty & Merenlender 2000). Given the importance of obtaining generally applicable and well-functioning indicator species sets, recent effort has been made to advance species selection methodologies with the advent of objective methods based on quantitative assessment of species' habitat preference (Renwick *et al.* 2012) or resource use (Butler *et al.* 2012, Wade *et al.* 2014). Whilst indicators based on these methods may produce comparable results to indicators based on expert opinion, particularly when there is high overlap in species inclusion (Butler *et al.* 2012, Renwick *et al.* 2012), their objective framework provides added rigour and opportunity for cross-comparison and benchmarking (Gregory *et al.* 2019).

Robust and informative biodiversity indicators also rely on well-designed monitoring programs to deliver representative data from which they are derived (Buckland & Johnston 2017). Although it is highly recommended to employ a random or stratified random sampling design (Gregory *et al.* 2004a, Schmeller *et al.* 2012), only about 30% of the large scale monitoring programs in Europe use such an approach to select monitoring sites (Schmeller *et al.* 2012). Even when random sampling is integrated into survey design through the identification of potential survey sites, spatial sampling bias may still be introduced by non-random selection of those sites by volunteer surveyors, geographic variation in the density of surveyed sampling units or bias towards sites closer to urban areas has been identified in monitoring schemes across many European countries, e.g. the Netherlands (van Turnhout *et al.* 2008, Boele *et al.* 2017), Austria (Teufelbauer *et al.* 2017), Sweden (Green *et al.* 2016), Denmark (Moshøj *et al.* 2017) and the

Czech Republic (Reif *et al.* 2008). A crucial question remains as to whether the surveyed areas are truly representative of the wider area (Butchart *et al.* 2010, European Bird Census Council 2016), because unweighted estimates of population changes may be biased if trends differ between geographic regions (van Turnhout *et al.* 2008, Morrison *et al.* 2013).

In this paper, we assess the degree of geographical and rural:urban bias in monitoring data used to construct a multi-species farmland bird indicator in Norway. We use our findings to inform the design of monitoring programs in countries with limited resources, for example a limited number of experts available to undertake bird surveys. Analyses are based on data from the Breeding Bird Survey (hereafter BBS) for Norway, which follows a random selection of routes (hereafter referred to as sites) from a defined national grid. We assume that any new monitoring scheme will apply a (stratified) random sampling design as previously recommended (Gregory *et al.* 2004a, Schmeller *et al.* 2012). We focus on the farmland bird community because populations of these species have shown severe declines in many areas (Fuller *et al.* 1995, Gregory *et al.* 2005, Butler *et al.* 2007), and thus multi-species farmland bird indices have broad geographic relevance. Using established methods to objectively select (Butler *et al.* 2012, Wade *et al.* 2014) and compute (Gregory *et al.* 2005) a multi-species indicator, we assess how the spatial configuration of monitoring sites can influence indicator trend estimates. Specifically, we compare the trends of the indicator when they are a) calculated using national monitoring data or data from a geographically restricted subset of sites, and b) calculated from monitoring data collected at sites within 10km to cities and other densely populated areas (hereafter Urban sites) or from sites >10km from the fringes of cities and other densely populated areas (hereafter Rural sites) as described by Statistics Norway (https://www.ssb.no/befolkning/statistikker/beftett/aar/2016-12-06). Finally, we compare this indicator with an existing farmland bird indicator comprising species selected using expert-knowledge (Husby & Kålås 2011). The results of this assessment will provide recommendations for design and implementation of new monitoring schemes, particularly when access to personnel or funding are limited and may restrict species coverage and/or the number and distribution of surveyed sites.

METHODS

Breeding bird survey

The Norwegian BBS data are collected from almost 500 sites which are randomly selected from among 1030 intersections of an 18km north-south by 18km east-west grid across the country (Husby & Kålås 2011, Lehikoinen *et al.* 2014, Lindström *et al.* 2015). The random selection of sites is stratified according to six regions: 1) east Norway; 2) south Norway; 3) west Norway; 4) central Norway; 5) Nordland and Troms counties; and, 6) Finnmark county. Each site consists of a route containing 20 point count stations situated 300m apart and forming a 1.5 × 1.5km square. In addition, nearly all observations of non-passerine birds (and a few pre-selected passerine birds) observed while moving between the counting points are recorded (Lehikoinen *et al.* 2014, Lindström *et al.* 2015). The number of observations for each sampling site is the sum of observed pair equivalents of birds at the counting points (5 min counting period at each point) and while moving between counting points (Kålås & Husby 2002). One pair is defined as an observation of either a male (most often singing), a female, a male and female observed together, or a parent with offspring (Koskimies & Väisänen 1991). For some sites, the number of counting points is less than 20 (but always ≥ 12) because of reduced availability (lakes, cliffs, rivers, etc.). Generally, counts are made between 23 May and 7 July (Lehikoinen *et al.* 2014, Lindström *et al.* 2015) when most bird species in the area are showing behaviours that enhance detectability, e.g. singing, searching for mates, alarm calling or other anti-predator behaviours (Kroodsma & Byers 1991, Catchpole & Slater 2008).

For this study, we used the complete national data (records from 223 of the 492 available sites across all six regions with at least one farmland bird species recorded, hereafter Norway) and data from east Norway (60 of 95 available sites, hereafter East) and central Norway (45 of 89 available sites, hereafter Central). These two latter regions have the longest time series available and relatively many sites positioned in farmland areas compared to the other regions. Within these two regions, sites were further classified as Urban (20 sites) or Rural (85 sites, Fig. 1). In Norway, about 3% of the land area is defined as farmland areas in use (9,800 km^2), of which 28% comprise corn and rape, and 68% grassland. In East, 5.7% of the land area is farmland area in use, and similarly 3.8% in the Central area. The farmland areas constitute 51% and 21% corn and rape, and 43% and 76% grassland in East and Central respectively (https://www.ssb.no/en/statbank/table/11342/). However, there are also other habitat types, e.g. open firm ground and wetlands, where we also can find farmland birds.

Selection of farmland indicator species

We used an objective, resource-use based selection algorithm (*SpecSel*, Butler *et al.* 2010b, Wade *et al.* 2014) to identify a representative and sensitive set of farmland bird indicator species. This approach draws on a matrix of species' ecological requirements covering components of diet, foraging habitat and nesting habitat, identifying combinations of species that, between them, exploit all resource types used by the wider community. For each indicator set size, and within this requirement for full resource coverage, the algorithm then identifies the species combination with the lowest average sensitivity score. Each species is scored for its reliance on farmland habitat to deliver resource requirements: major (scored as 1), moderate (2) or minor (3), with sensitivity to environmental change in farmland calculated as the number of resources it uses multiplied by its reliance score, with higher scores therefore attributed to less sensitive species (Butler *et al.* 2010b, Wade *et al.* 2014). We then identified the indicator set size with the lowest overall average sensitivity score (i.e. the most sensitive) and used this species combination for our farmland bird indicator.

From a community of 28 farmland bird species in Norway (defined as species with ≥80% of estimated population using farmland as breeding habitats), 17 are sufficiently widespread and abundant (observed in ≥50 sites in the full dataset, i.e. Norway; Kålås *et al.* 2014) to calculate population trends from Norwegian BBS data (Table S1). The resource requirements of each of these species in Norway was assessed based on existing literature (Haftorn, 1971, Cramp 1985, 1988, 1992, Cramp & Perrins 1993, 1994a, 1994b, Cramp & Simmons 1977, 1980, 1983, Husby & Kålås 2011), the authors' own experience and feedback from regional leaders in the Norwegian BBS. We assessed use of four potential diet components (below-ground invertebrates, above-ground invertebrates, plant material, seeds) in each of three potential foraging habitats (cropped area, margins, hedgerow) and used three potential nesting locations (cropped area, margins, hedgerow). Note that, contrary to previous applications of this approach elsewhere (Butler *et al.* 2010b, Wade *et al.* 2014, Teufelbauer *et al.* 2017), we only collated data on summer resource use as few species remain on farmland habitats in Norway over winter. We excluded vertebrate prey as a potential diet component as the suite of farmland bird species did not include any predatory species.

Data analysis

For each species selected for inclusion in the indicator set, we calculated population indices for all sites in Norway, Central, East, Central+East combined, Central+East Urban, and Central+East Rural. Index calculations were based on loglinear regression using the rtrim-package (Bogaart *et al.* 2016) in R (version 3.5.1, R Core Team 2018), with random effects for sites and correction for over dispersion (setting over dispersion as TRUE in rtrim models). Correction for serial autocorrelation (setting autocorrelation as TRUE in rtrim models) was included only for one model (*Alauda arvensis* in Urban), since all other models had negative correlation coefficients or low coefficients (rho < 0.2). Models were fitted with year-dependent effects except for two models (*Alauda arvensis* in Central had data from only 4 sites, *Numenius arquata* in Urban had data from only 5 sites and had an observation of zero or missing data for 2010 - rtrim requires at least one observation >0 to provide an estimate), which were fitted with linear time-effects. Calculation of the multi-species indicator followed the method suggested by Soldaat *et al.* (2017), which uses Monte Carlo simulations in R to calculate trends and standard errors. This accounts for sampling error in the indicator and allows testing of differences between trend lines (MSI-tool, Soldaat *et al.* 2017).

For each data subset, we extracted geometric means of the species-specific annual indices obtained from rtrim. We defined the first monitoring year as the base year with the index set to 100 and the standard error set to zero for each species. The indices for the remaining years were expressed as percentages of the base year, and the standard error was a function of the variance in the specific year and the base year. In the resulting composite MSI and smoothed trend for our farmland birds, every species is weighted equally (Gregory *et al.* 2004b). The simulation procedure was based on the approximately log-normal distribution of the standard errors of index values. For each data subset, the yearly index for each species was drawn 1000 times from a normal distribution $N(\mu, \sigma)$, where μ = the natural logarithm of the index and σ = the standard error of the index on the log-scale. The standard error of the index on the log scale was calculated by the Delta-method (Agresti 1990). The annual multi-species indicator (MSI) and smoothed linear trend were calculated for each simulation.

We set the MSI-value to 100 for the start year, and the trend values by Monte Carlo simulations, so the MSI-values deviated slightly from the standardized trend values (Soldaat *et al.* 2017). After simulation, the mean and standard error of each simulated multi-species indicator was calculated and back-transformed to the index scale. Classification of trends followed the procedure used in TRIM software for analysis of biological time series data (Pannekoek & van Strien 2005). In addition, we obtained the overall trends from 2007 to 2016 as the ordinary least squares (OLS) estimator of the slope parameter, presented as the slope of the regression line with intercept (additive trend) and as the slope of the regression line forced through the base time point (year 2007; multiplicative trend, Pannekoek *et al.* 2018).

We then used a simulation procedure to examine the impact of monitoring site clustering, either geographically or along an rural:urban gradient, on indicator characteristics. We chose to run the simulations with 300 sites to stabilize the data frames created at each iteration. Initial simulation attempts using the average number of sites with data across all species within each geographical region (52 sites) failed to run due to a low number of observations recorded in the iterations of some region-species combinations. This choice might artificially lower the confidence intervals in the models, and we present careful interpretations based on the confidence intervals, choosing to focus on general patterns. Firstly, for each of the 1000 iterations, we drew 300 sites with replacement from either the Norway, Central, East or Central+East datasets, and ran species-specific TRIM-models on the corresponding monitoring data to calculate corresponding annual MSI values, MSI-standard deviation, trends and confidence limits. We evaluated trend similarity between regions based on the average MSI, standard deviations, trends and their confidence limits across iterations. Secondly, for each of the 1000 iterations, we drew 300 sites with replacement from the Central+East region sites, ensuring selected sites included i) 100% rural sites, ii) 75% rural and 25% urban sites, iii) 50% rural and urban sites, iv) 25% rural and 75% urban sites, or v) 100% urban sites. We used the combined Central+East region data for these simulations because sites outside Central or East regions were not defined as rural or urban, and sampling based on Central or East regions alone did not provide stable or reliable results as judged by confidence intervals that were undetermined, or that exceeded the boundaries of expected variation. Again, we ran species-specific TRIM-models on monitoring data from each selected set of sites and calculated the corresponding annual MSI values, MSI-standard deviation, trends and confidence limits around the trends accordingly. We evaluated differences in trends along the rural-urban site composition gradient from the average MSI, standard deviations, trends and their confidence limits. These trends were also compared to those calculated for Norway and Central+East regions in the first simulation exercise. Note that, due to the nature of species distributions across sites defined as rural or urban, the simulation procedure sometimes led to some species not being present in all data sets for all iterations.

We then compared the multi-species indicator we developed using an objective species selection with an existing Farmland Indicator Index based on species selected using expert knowledge. This comprised the 7 species also selected in our indicator set plus *Saxicola rubetra* (see Results, Husby & Kålås 2011). We ran species-specific rtrim analysis for *S. rubetra* for Norway, Central, East and Central+East and integrated these with the indices previously calculated for the other 7 species to generate MSI-values and smoothed trends for each geographical subset of sites. In addition, we calculated indicator precision as the average difference between the annual 95% confidence intervals over 10 years, following Butler *et al*. (2012) and assessed the influence of species composition on precision using linear regressions (lm-function from R base package).

RESULTS

Application of the *SpecSel* algorithm identified an indicator set containing seven species as the most sensitive combination. This set included Eurasian Curlew *Numenius arquata*, Northern Lapwing *Vanellus vanellus*, Eurasian Skylark *Alauda arvensis*, Common Starling *Sturnus vulgaris*, Yellowhammer *Emberiza citrinella*, White Wagtail *Motacilla alba* and Barn Swallow *Hirundo rustica*. Four of these species (Eurasian Curlew, Eurasian Skylark, Yellowhammer and Common Starling) have shown moderate declines across Norway, whilst Northern Lapwing has undergone steep declines. Trends of White Wagtail and Barn Swallow are uncertain (Table S2). Species-specific trends for the regions and for urban or rural sites mostly showed similar patterns to their national trends, albeit with some exceptions (Table S2). National, regional and rural:urban gradient multi-species indicators were derived from the population trends of these seven species.

MSIs and trends based on original data

The Norway multi-species indicator showed a significant decline of 6.1% per year from 2007 to 2016 (Fig. S1). The regional and Rural indicators showed similar moderate declines, whilst the Urban indicator was classified as showing a steep decline over this time period (Table 1). Norway trends were classified as showing moderate decline in most years (Table S3), whilst annual trends for the two regions, and for both rural and urban sites, were classified as showing moderate population declines during the first years of monitoring (3-5 years depending on region), but as being stable or uncertain in more recent years (Table S3).

MSIs and trends based on simulated data

Geographical clustering of survey sites did not significantly influence indicator trends, with comparable trends and MSI values between Norway, Central, East, and Central+East (Table 1, Fig. S2). Clustering of survey sites along the urban:rural gradient did influence indicator trends, with the simulated Central+East indicator showing an increasingly steep decline as the proportion of Urban sites contributing data increased (Fig. 2). However, whilst rural sites made up 81% (85 sites) of survey sites across the Central and East regions, this simulated indicator only fell significantly below the observed Central+East indicator when the proportion of rural sites contributing data was below 15% (Fig. S2, Fig. S4, Appendix S1). Note that this simulated indicator deviated from the full Norway indicator at a slightly higher ratio of rural:urban sites (25% rural sites; Fig. S4, Table S4)

The bird indicator developed in this paper, using an objective species-selection process, contains 7 of the 8 species in the existing indicator set selected using expert knowledge (the 8th being *S. rubetra*, Fig. 3a, Tables S5, S6 and S7). There was no significant difference in the trends of these two farmland indicators, but the 8-species, expert-knowledge based indicator reported less negative changes in populations, with the difference in trend values between the two indicators steadily increasing between 2007 and 2013 and remaining consistent since 2012/2013.

In both the 7-species and 8-species sets, estimated precision depended on region (7-species dataset: $F_{5,54} = 15.27$, $P < 0.001$, 8-species set: $F_{5,54} = 11.79$, $P < 0.001$). Precision was greatest (lowest value) when based on data from the larger regions (Norway and Central+East), whilst indicators using data from the smaller regions, or from Rural and Urban sites within the Central+East region, were substantially less precise (Fig. 3b). Precision estimates did not differ between the two indicator sets (region: $F_{5,108} = 15.03$, $P < 0.001$, set size: $F_{1,108} = 0.03$, $P = 0.86$, region*set size: $F_{5,108} = 0.12$, $P = 0.99$).

DISCUSSION

Our multispecies farmland bird index revealed population declines between 2007 and 2016 across Norway, and in each region and rural:urban gradient subsamples. For Norway, Central, East, Central+East and Rural sites, declines were all classified as 'moderate', although declines have been significantly more negative in the two regions than across Norway as a whole, but significantly less negative in Rural sites. The exception was the index based on monitoring data from sites within 10km of urban areas, where declines in farmland bird populations over this time period were classified as ‘steep’.

The objective selection of species using the *SpecSel* algorithm resulted in a farmland bird indicator containing seven species. For sequentially increasing set sizes, the *SpecSel* algorithm identifies the combination of species with the lowest average sensitivity score (i.e. the most sensitive) that, between them, use the full range of resource exploited by the wider community. The set of seven species identified here was the set with lowest average sensitivity score across all potential set sizes (Wade *et al.* 2014). This is one fewer than included in the farmland bird indicator currently used by the Norwegian government to assess national targets on biodiversity (http://www.environment.no/goals/1.-biodiversity/target-1.1/), for which species selection is based on expert knowledge (Husby & Kålås 2011). In addition to the seven species included in our indicator, the expert-determined list includes Whinchat. Interestingly, the optimal set containing eight species identified by *SpecSel* included Whinchat in addition to the species in the seven-species indicator set (unpubl. data). Species inclusion in the indicators therefore seem to be relatively consistent between these two selection methods, which is in agreement with other comparisons of species selection methods for ecological indicators (Renwick *et al.* 2012). Including Whinchat data in the multispecies indicator resulted in similar trend values compared to the 7-species indicator and the main inferences from the indicators are the same. However, the 8-species indicator consistently provided MSI values and trend classifications that indicated lower declines in farmland birds than the 7-species indicator developed in this paper. This was because of the uncertain to stable trend classes of Whinchats within the regions (Tables S6 and S7). Indicators including larger sets of species may produce indices with higher precision, especially if the additional species are generalist species that are more widespread and/or have more stable population dynamics (Butler *et al.* 2012). This can come at a price of reduced indicator sensitivity, especially when species sets are large, and may produce an indicator that performs sub-optimally (Lindenmayer *et al.* 2000). However, the difference in indicator set size was low in our study, and there was no difference in precision between the two indicator sets (Fig. 3). During the short time window for which we have data in this study, the performance of the 7-species and 8-species indicators seem to be comparable and there is no apparent evidence of biases caused by the size of the species sets.

Farmland bird declines in Norway correspond with the strong declines of these species reported across Europe and North-America (Gregory *et al.* 2005*,* Reif 2013*,* Stanton *et al.* 2018). These declines have been largely driven by agricultural intensification (Chamberlain & Fuller 2001*,* Reif 2013*,* Stanton *et al.* 2018), but it is likely that the negative effects of intensification on bird populations are exacerbated by climate change (Kleijn *et al.* 2010, Jørgensen *et al.* 2016*,* Santangeli *et al.* 2018). In Norway, the rate of decline in farmland bird populations was greatest between 2007 and 2011 and then stabilized to some extent in subsequent years, particularly in Rural sites and the Central region. However, farmland bird populations in Norway are expected to undergo further declines over the coming decades in response to predicted land-use changes (Scholefield *et al.* 2011) and to climate change effects on, for example, community composition (Forsgren *et al.* 2015)*,* and the transition from stable to uncertain trends, and apparent worsening in the rate of decline in the last couple of years in some of the route subsamples support this (Table S1*,* Fig. S2).

The Urban site indicator suggests that farmland bird populations in areas within 10km of dense human populations have fared worse than elsewhere in Norway, and Central+East indicators derived from sites including 85% Urban sites or more showed significantly greater population declines than the observed trend for Central+East (Fig. S3, Fig. S5). Compared with the Norway indicator, indicators that included 25%-50% Urban sites showed significantly more negative population trends (Fig. S4). Some individual species disappear with urbanization while others increase in abundance (Blair, 2004). Urban areas may include more unsuitable or lower quality habitat for farmland specialists, or result in higher competition with generalist species that are less affected by urbanization (Krauss *et al.* 2003, Devictor *et al.* 2007). We have found that dense urban areas are gradually becoming more unsuitable for most farmland bird species, perhaps due to reduction of suitable habitats or more intensive farming near urban areas (Hendershot *et al*. 2020). Even at low levels of urbanization (25% urbanization), adjacent farmland community composition is found to differ considerably from undisturbed communities (0% urbanization, Filippi-Codaccioni *et al.* 2008), supporting our hypothesis of lower quality habitats close to urban areas causing stronger population declines. Since the *SpecSel* algorithm used to select species in this paper optimally selects for specialised species, the species set could be expected to be more severely affected by habitat degradation (Filippi-Codaccioni *et al.* 2008*,* Krauss *et al.* 2003) and land use changes and disturbance (Devictor *et al.* 2007*,* Schweiger *et al.* 2007). These processes may affect farmland near urban areas more negatively than rural farmland (Mason 2006, Gundersen *et al.* 2017). Conversely, the weaker declines in rural sites suggest either a buffered response of farmland birds to widespread detrimental changes or that some changes are occurring disproportionately less in these areas. Similarly, the stronger declines observed in Central and East regions compared with the Norway trend may relate to the relative distribution of habitats of different quality across regions of Norway, or variation in the extent or strength of detrimental environmental changes, underpinned for example by regional differences in land-use policies (Hanzelka *et al.* 2015).

The proportions of rural areas (including farmland, open firm ground and wetlands) in the East and Central regions are about 24% and 50% respectively, whilst the proportions of urban areas in East and Central regions are 9% and 2% respectively, suggesting that availability of good quality habitat for farmland birds may indeed differ between regions (Adapted from Statistics Norway, https://www.ssb.no/en/natur-og-miljo/statistikker/arealstat – Table 2). Compared with these numbers, the percentage of sites defined as Urban and Rural in the breeding bird survey dataset deviated substantially. This could be due to how urban and rural sites were defined here (based on distance to cities and other densely populated areas) compared with the percentage cover of land cover types. However, when only considering these two land use types, reflecting the definition of Urban and Rural survey sites used here, the average coverage of urban and rural land for Central+East (29% and 71%, respectively) differs markedly from the observed proportion of Urban and Rural sites in Central+East (19% and 81%, respectively).

Spatial variation in population trends between regions has previously been reported for farmland birds in Sweden (Wretenberg *et al.* 2007) and the UK (Harrison *et al.* 2014, Massimino *et al.* 2015) and emphasizes the importance of a random or regular sampling scheme for monitoring bird population trends (Gregory *et al.* 2004a). However, density variation in sampling sites across countries is common and may lead to biases in trend estimates if this leads to unequal sampling across the range of environmental or land-use changes, or of habitats of different quality (Reif *et al.* 2008, van Turnhout *et al.* 2008, Wellicome *et al.* 2014, Teufelbauer *et al.* 2017). Some of the potential biases caused by density variation in sampling effort can be corrected for statistically (van Turnhout *et al.* 2008), but the optimal solution is to avoid such biases by implementing a stratified random design (Gregory *et al.* 2004a). In situations where spreading sampling sites across the country is logistically difficult, some parts might have so few routes investigated that even statistically weighting is impossible (Rosenberg *et al.* 2017). Reliable trend estimates may then be obtainable by sampling smaller regions if adhering to a stratified, random sampling regime where all relevant habitat types, land-use policies and other factors that may lead to estimate biases are represented in similar proportions within the sampled region as within the country. However, sampling regimes where sites are concentrated around urban areas, for example, should be treated with care as such trend estimates may differ considerably from population changes at national levels.

Conclusions

Concentrating sampling sites around urban areas, where it may be more likely to find qualified volunteers for sampling, should be avoided as trend estimates derived from data collected at these sites can be considerably different from national trends. However, sampling a smaller region of a country may provide trends of similar direction and magnitude to national-scale trends if survey site distribution is stratified according to national availability of habitat types or other relevant factors that may bias trend estimates. Regardless, it is important to keep in mind that spatial differences in indicator trends provide information relevant for determining conservation priorities (e.g., Massimino *et al.* 2015). Therefore, we follow recommendations of Gregory et al. (2004a) and others (Bibby *et al.* 1992*,* Voříšek *et al.* 2008), and suggest that a stratified sampling design across the whole study area/country (e.g. habitat, geography, human density) will increase the probability of obtaining a representative sample and provide the most accurate trend estimates.

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DATA AVAILABILITY

The Norwegian BBS data is available from the Norwegian Institute for Nature Research (NINA), Trondheim, Norway.

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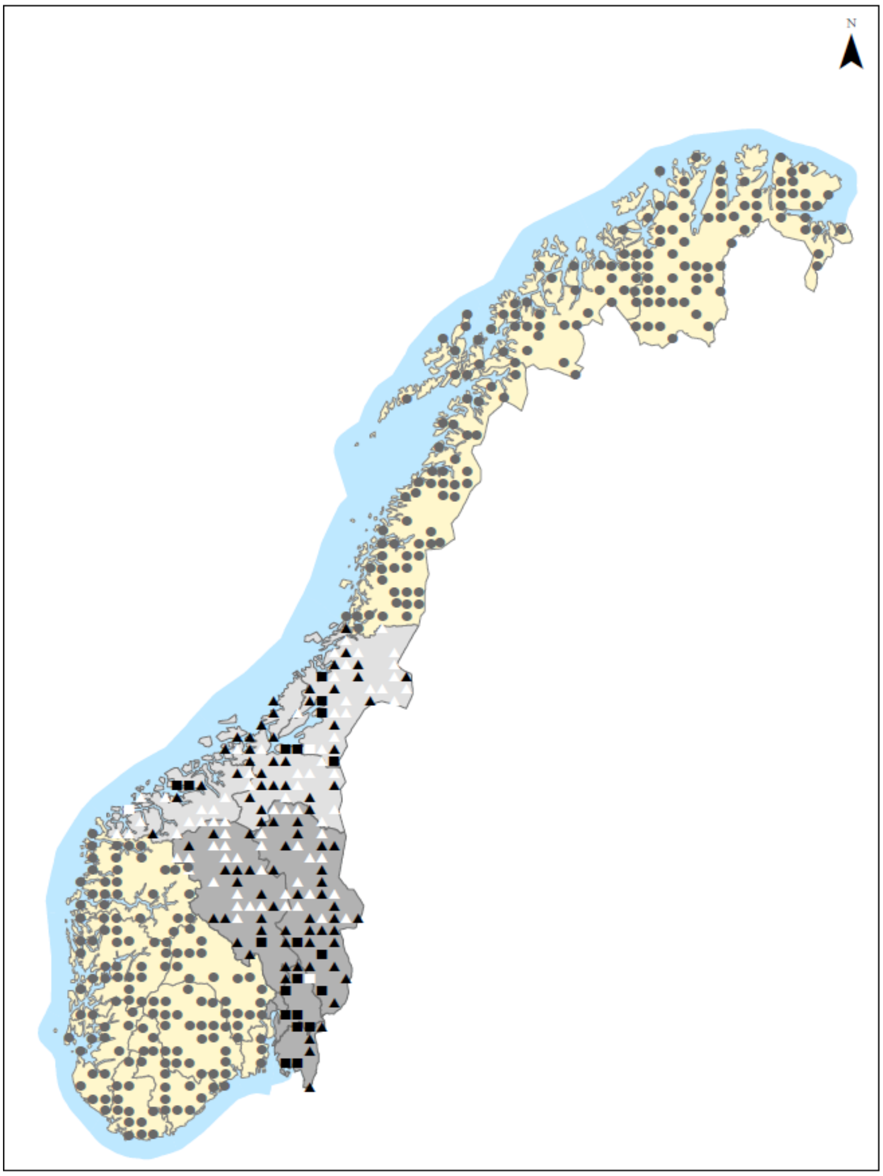

Figure 1. Map of active Breeding Bird Survey sites in Norway with Central (light grey area) and East (dark grey area) regions highlighted. Within Central and East, squares represent Urban sites and triangles represent Rural sites, black and white symbols represent respectively records or no records of the farmland birds included in our new farmland bird indicator. Dark grey circles represent all monitoring sites outside the Central and East regions. Scale 1:6500000. Map data from Norge Digitalt.

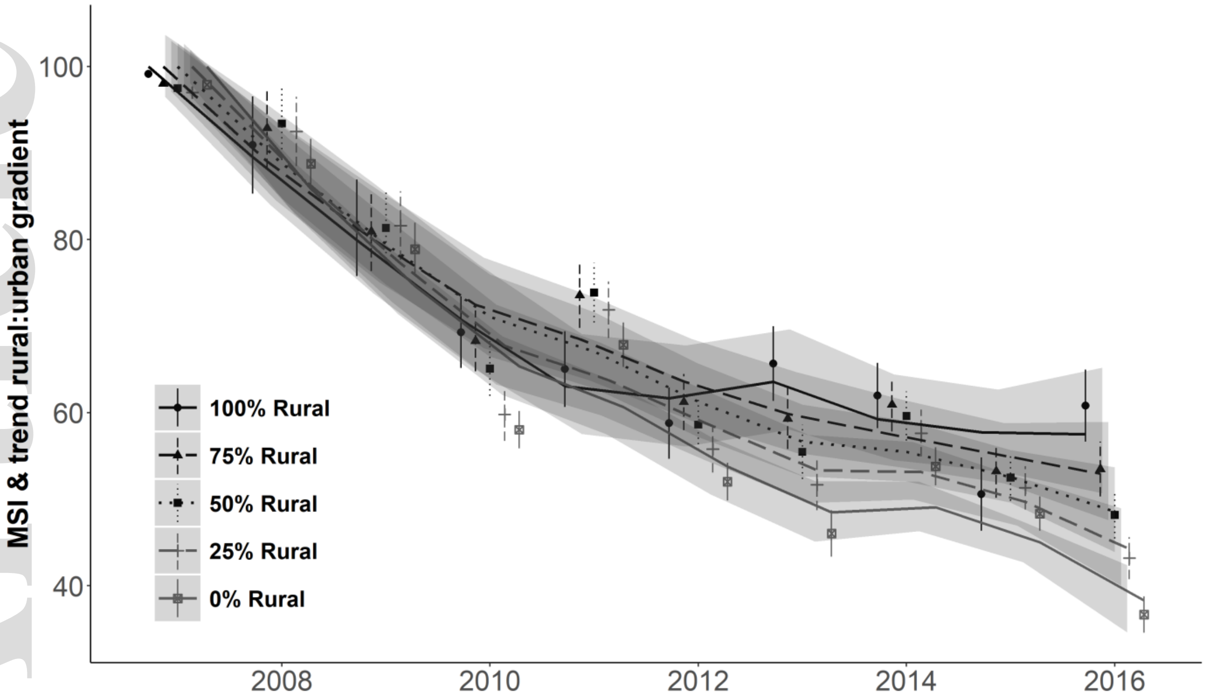

Figure 2. The influence of spatial clustering of Central+East survey sites along a rural:urban gradient on the average MSI values ± average sd, and smoothed trend line values with average lower and upper confidence limits.

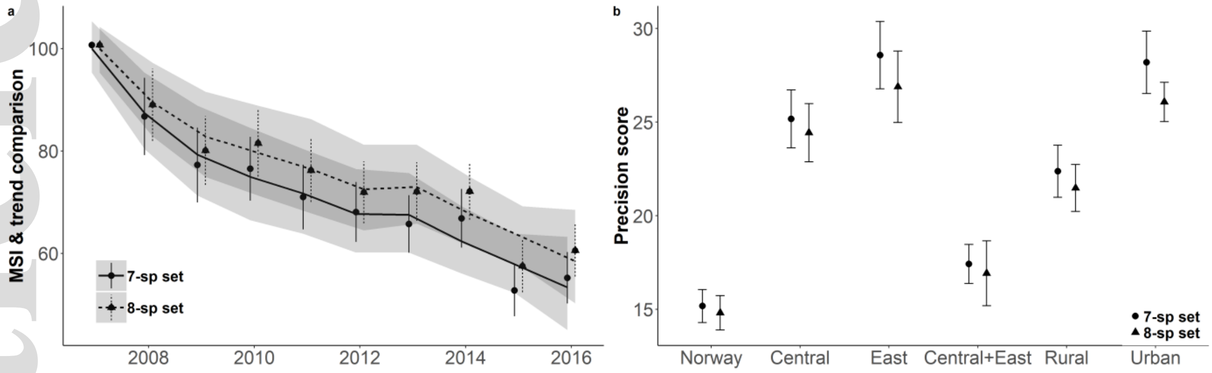

Figure 3. Comparison of a) MSI and trend values and b) precision estimates of the 7-species indicator based on the *SpecSel* algorithm and the 8-species indicator based on expert knowledge for the Norway region.

Table 1. Additive (slope parameter of regression line with intercept) and multiplicative (slope parameter of regression line forced through the base time point of 2007) trends and trend classes for all data subsets/regions (Norway, Central, East, Central+East combined, Urban (< 10km from city or town), and Rural). Both representations of the slope parameter are bounded between 0 and 1.

Region/subset	N sites	Additive trend ± SD	Multiplicative trend ± SD	Trend class
Norway	223	-0.062 ± 0.009	0.940 ± 0.008	Moderate decline
Central	45	-0.063 ± 0.014	0.939 ± 0.013	Moderate decline
East	60	-0.069 ± 0.017	0.933 ± 0.016	Moderate decline
Central+East	105	-0.066 ± 0.010	0.936 ± 0.010	Moderate decline
Rural	85	-0.060 ± 0.014	0.942 ± 0.013	Moderate decline
Urban	20	-0.094 ± 0.019	0.911 ± 0.018	Steep decline