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RESEARCH ARTICLE

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Remote sensing reveals scale-specific effects of forage crop mowing and landscape structure on a declining farmland bird

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Abstract

1. The effectiveness of agri-environment schemes (AESs), the largest conservation-related expenditure for farmland biodiversity conservation within the European Union, is often compromised by a limited spatial scale of implementation. We focused on multiannual forage crops, a surrogate habitat for grassland birds, to assess the scale-dependent effects of mowing timing and frequency on the local population size of an iconic species, the skylark (*Alauda arvensis*). While there is much evidence for a negative impact of in-field mowing activities on grassland birds, whether such effects occur also at broader spatial scales is largely unknown.
2. We surveyed breeding skylarks in the Po Plain (northern Italy) to determine (1) the association between landscape composition/configuration and abundance and (2) how abundance is affected by forage crop mowing timing and frequency. We addressed both questions through scale optimisation, identifying the most influential spatial scales for each covariate. Forage crop mowing timing was assessed through a novel remote sensing algorithm based on high-resolution Sentinel-2 satellite images.
3. We observed a strong scale dependence on the importance of different habitats in determining skylark abundance. Abundance increased with an increasing cover of forage crops locally (200 m) and of winter crops at a landscape scale (2600 m), suggesting that the species is favoured by heterogeneous agroecosystems. Locally (150–350 m), skylarks were more abundant when crops were aggregated, being negatively impacted by crop fragmentation caused by urbanization and by semi-natural habitats.

4. At the landscape scale (1150 m), the timing of mowing was consistent across years, with early-mown areas supporting fewer skylarks. This is probably because, over longer temporal scales, early-mown forage patches have limited or null productivity, eventually limiting local population size.
5. *Synthesis and applications*. We provide a new perspective on the overarching influence of spatial scale in driving the abundance of a declining farmland bird species, supporting the urgency of designing landscape scale-effective AESs. This should be framed within the new EU Common Agricultural Policy reform and operated by farmer collectives, whereby management interventions should be monitored by state-of-the-art remote sensing techniques. These results suggest that implementing scale-optimized AESs could be crucial for effective farmland biodiversity conservation.

KEYWORDS

agri-environment schemes, agroecosystem, *Alauda arvensis*, CAP, crop rotation, Sentinel satellite, skylark

1 | INTRODUCTION

Natural and seminatural grasslands harbour an extraordinarily rich biodiversity, which has been maintained by low-intensity farming practices for millennia (Pykälä, 2000; Veen et al., 2009). Since the second half of the last century, grassland management practices have become more intensive, or, when no longer economically sustainable, have ceased, leading to a profound crisis in grassland-dwelling plants and animals (Douglas et al., 2023; MacDonald et al., 2000). It has been estimated that a net loss of 190 million grassland birds occurred in the European Union between 1980 and 2017 (Gregory et al., 2023). This decline has been exacerbated by the introduction of the EU Common Agricultural Policy (CAP), which has supported both farming intensification and land abandonment, with severe impacts on grassland biodiversity (Assandri et al., 2019a; Pe'er et al., 2014). However, CAP has also introduced agri-environment schemes (AESs), that is policy mechanisms in which farmers are financially incentivized to adopt environmentally beneficial measures. AESs represent the highest conservation-related expenditure in the EU (Batáry et al., 2015), yet whether their monetary cost translates into effective biodiversity conservation in agroecosystems has been questioned. A large amount of literature has shown that AES effectiveness depends on whether measures are targeted to the specific conservation issue and on the spatial scale of implementation (Kleijn et al., 2006; McCracken et al., 2015).

Across European landscapes, natural and seminatural grasslands have often been replaced by intensive annual crops (Lambin et al., 2001; Pelletier & Tyedmers, 2010). In intensive cereal croplands, multiannual forage crops (i.e. cultivation of plants grown specifically to provide feed for livestock) can be considered the best possible surrogate for grassland habitats because they are subject to a rotation period of 3–4 years (or even longer) and agricultural practices are generally less intensive than those applied in annual crops (Bretagnolle et al., 2019; Buckingham et al., 2015). As a result, these habitats are attractive for several declining grassland bird species (Assandri et al., 2023; Buckingham et al., 2015). However, multiple and early-season harvests have been shown to cause substantial losses of nests and fledglings, often transforming forage crops into an ecological trap (Buckingham et al., 2015; Green, 1996).

Although there is substantial evidence for the negative effects of in-field mowing practices on grassland-dwelling birds (Assandri et al., 2019b; Brambilla et al., 2021; Strebel et al., 2015), there is still a dearth of data on how mowing practices affect bird abundance at a broad spatial extent (i.e. landscape; Canonne et al., 2024). This is largely due to the technical restrictions in mapping mowing events at large scales (e.g. regional). From a conservation and policy perspective, this represents an implementation gap, as the few studies that have addressed AES biodiversity conservation effectiveness at broader-than-farm scales have often found that biodiversity-friendly management practices are delivered by AESs at spatial scales that are too small to achieve population-level effects (Pe'er et al., 2019; Sharps et al., 2023; Siriwardena, 2010).

In this study, we focused on the Eurasian skylark *Alauda arvensis*, a ground-nesting grassland bird species, which has undergone large population declines in European farmland regions (BirdLife International, 2017; Burfield et al., 2023), to unravel potential landscape-scale effects of forage crop mowing on its abundance. We relied on standardized skylark counts conducted in an intensive agricultural area mostly dedicated to the production of forage crops (southeastern Po Plain, northern Italy) to first determine the association between landscape composition/configuration at different spatial scales and skylark abundance; secondly, we investigated how and at which spatial scale skylark abundance was affected by forage mowing timing and frequency. We predicted that (1) abundance should be positively associated with the cover of forage crops, as this cultivation represents a grassland habitat surrogate in intensive agroecosystems, and (2) abundance should be reduced with earlier cut dates and more frequent mowing events. The latter may occur because consistently early and more frequent forage cuts reduce survival and productivity via increased direct mortality and nest loss (Buckingham et al., 2015; Wilson et al., 1997), which can result in lower abundance of the species. The scale at which this mechanism is likely to occur is difficult to predict given the limited knowledge available to date.

We determined the timing and frequency of mowing by leveraging a novel remote sensing methodology integrating high-resolution satellite images from Sentinel-2 (Andreatta et al., 2022). This allowed us to estimate forage crop mowing parameters at increasing spatial extents with respect to the bird sampling location, from 150 to 3000 m. Our ultimate goal was to disentangle the spatial scale(s) at which a given farming practice (i.e. mowing of forage crops) is most influential in driving spatial variation in the abundance of a declining farmland species. From a conservation and landscape management perspective, our results might be applied to develop AESs at a spatial scale tailored to the ecology of the target species.

2 | MATERIALS AND METHODS

2.1 | Target species

The skylark favours natural and seminatural grasslands as a breeding habitat, although arable lands harbour a considerable part of the species' European population (Chamberlain & Siriwardena, 2000; Copland et al., 2012). Managed grasslands and forage crops are actively selected by the species, particularly in the absence of natural grassland (Püttmanns, Böttges, et al., 2022; Wilson et al., 1997). In common with several other ground-nesting farmland birds (Guerrero et al., 2012; Li et al., 2023), the skylark has shown a dramatic decline at the European scale since the second half of the 20th Century and it is now considered a species of European Conservation Concern (Burfield et al., 2023; https://pecbms.info/trends-and-indicators/species-trends/species/alauda-arvensis/). In Italy, it declined by 53.6% between 2000 and 2023 (Rete Rurale Nazionale & Lipu, 2024a).

2.2 | Study area and design

This study was performed in the southeastern part of the Po Plain (Lombardy and Emilia-Romagna regions, northern Italy), a densely inhabited and intensively cultivated region (Falcucci et al., 2007). In the study area, forage crops are well represented, as they are fundamental to the production of the *Parmigiano Reggiano* PDO cheese, whose production specification rules do not allow silage and require that at least 75% of the forage provided to cattle must be cultivated in the area (Battini et al., 2016; Mantovi et al., 2015).

Our study was designed to understand both the importance of forage crops and their management for the skylark. To capture a wide gradient of forage crop cover, we overlaid a 1-km grid over three subareas of the southeastern plain (between the city of Cremona and the Adriatic Sea) resulting in 2467 1-km^2 landscape cells initially included in the study (Figure 1, see also Assandri et al., 2023 for further details). One square kilometre is a scale previously shown to be relevant for nest site selection in ground-nesting passerines (Pickett & Siriwardena, 2011). In addition, at this scale, several territories of skylark (given their average size; Donald & Harris, 2010) are likely to occur, resulting in an abundance gradient suitable for the analysis.

For each cell, we calculated the cover of each land use/land cover (LULC) category (see Supporting Information for additional details). Within this area, we retained the 2301 landscape cells with more than 33% of agricultural LULC, and within them, we randomly selected 120 landscape cells according to four strata (30 cells in each stratum) defined based on the cover of forage crops within the cell: high (≥55%), medium (≥30 to <55%), low (≥5% to <30%) and very low (<5%).

2.3 | Skylark survey

In each of the 120 selected landscape cells, we undertook a point count survey located as close as possible to the centroid of the cell (mean distance between two nearest-neighbour point counts: 1304 m; range: 615–4234 m). We counted skylarks at each point during three count sessions during the 2021 breeding season (7–20 April; 12–20 May; and 12–18 June) within a 150 m radius (area: 7.07 ha), which was previously considered to be adequate to sample open-habitat bird species in the same area (Assandri et al., 2023). During each of these count sessions, we surveyed 12 point counts each day (hereafter 'point cluster'), from half an hour after sunrise and for the subsequent 4 h, on days with favourable weather (Bibby, 2000). Among periods, the point count sampling sequence was reversed within clusters. Each point count lasted 10 min (Bibby, 2000) during which we mapped all the initial contacts with every individual skylark on updated high-resolution aerial photographs (scale 1:2000 m). We derived a minimum number of skylark territories for each point count based on simultaneous contacts and reproductive cues and finally retained the maximum territory count among the three repetitions (Assandri et al., 2018; Broyer et al., 2012). Neither specific permissions nor ethical approval were required to conduct this study.

2.4 | Landscape and forage mowing variables

We measured seven landscape covariates in circular landscapes around each point count, starting from a radius of 150 m and ending at 3000 m, with a 50 m incremental increase in the radius ($n = 58$ spatial scales). Each covariate was thus extracted at 58 spatial scales (see Guttery et al. (2017) for a comparable approach). For each landscape of increasing size, we assessed the percentage cover of five LULC categories based on the 2021 release of the LULC map (see Section 2.2): (1) forage crops (mostly

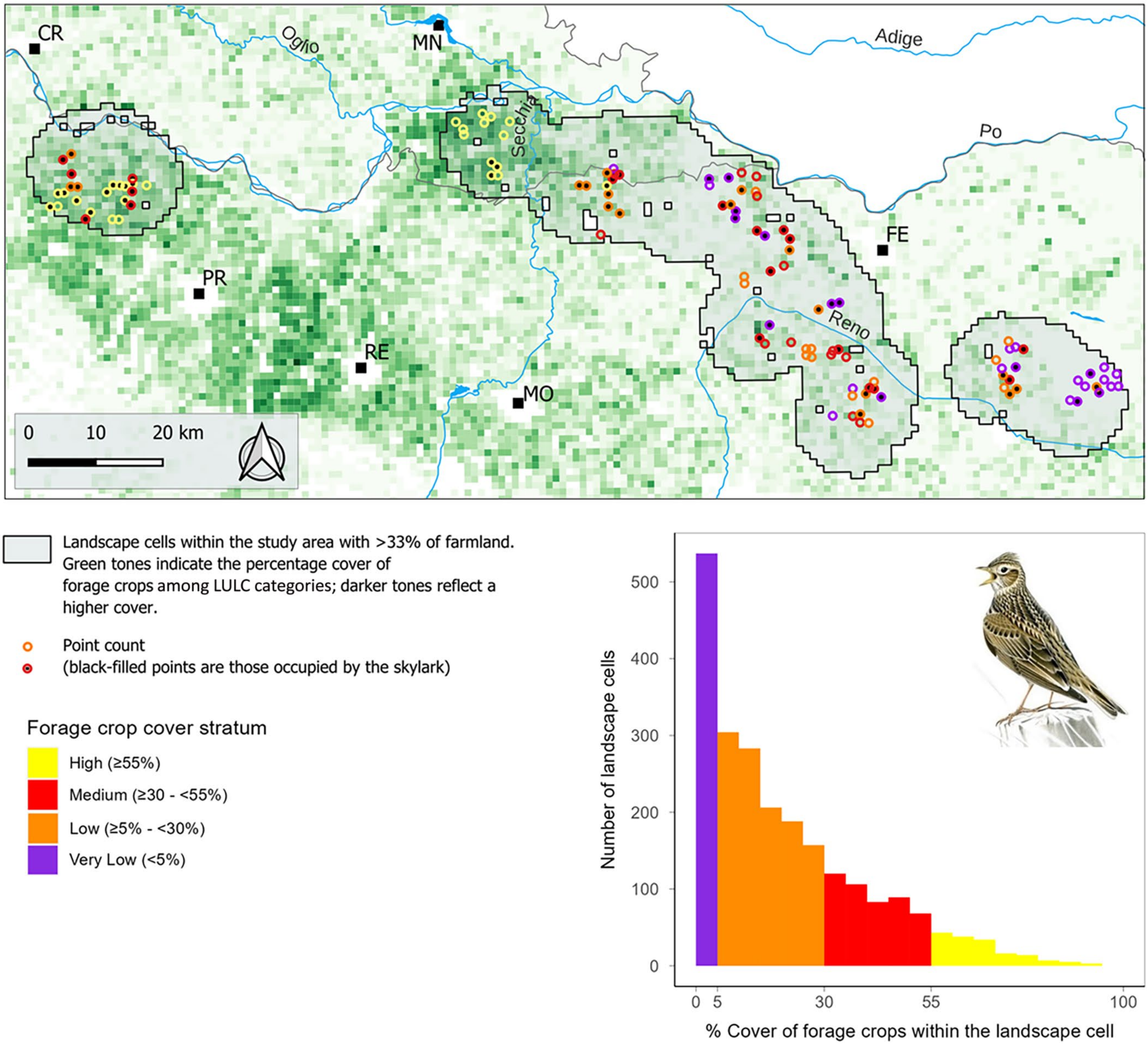

FIGURE 1 Map of the study area and sampling design. The top panel shows the location of the point counts within the study area. The circle colour represents the forage cover within the 1-km cell (the overall distribution in the study area is shown in the bottom right bar plot; see legend for colour interpretation). Black-filled dots represent those at which at least one skylark territory occurred. The main cities in the area are highlighted with black squares (CR = Cremona; PR = Parma; RE = Reggio Emilia; MN = Mantua; MO = Modena; FE = Ferrara). Original artwork by Lorenzo Starnini (© ISPRA).

alfalfa *Medicago sativa*, cultivated for 4–5 years with up to four to six mowing events per season, then converted into winter/summer crops for at least 5–6 years before the cycle restarts); (2) winter crops (mostly cereals sown at the end of autumn and harvested at the beginning of the following summer, for example wheat and barley); (3) summer crops (sown at the end of winter/beginning of spring and harvested in the same summer, mostly maize and soybean); (4) seminatural habitats (mostly woodlots and wetlands, including ditches/channels); and (5) built-up areas, including roads (Figure S1a). In addition, we calculated an index of landscape compositional heterogeneity (Shannon landscape diversity index; hereafter 'SHDI') and an index of configurational heterogeneity (Aggregation Index, hereafter 'AI') using the R package *landscapemetrics* (Hesselbarth et al., 2019). At large scales, circular landscapes were partially overlapping; however, according to Zuckerberg et al. (2020), this should not be regarded as a violation of statistical independence.

We mapped three forage mowing parameters for 2021 at each point count and for each scale (150–3000 m; Figure S1b) using the mowing detection algorithm developed by Andreatta et al. (2022). This algorithm profited from the high spatial and temporal resolution of Sentinel-2 (S2) imagery (respectively up to 10 m and 2–5 days) to effectively detect mowing events. Since the method was initially developed to map the frequency of mowing in montane grassland rather than the first cut date in lowland forage crops, we assessed its estimation accuracy in the context of our study area and investigated whether any adjustments were needed. To this end, we developed a reference dataset of first cut dates based on visual inspection of PlanetScope imagery (details in Supporting Information 1.2). The reference dataset indicated that, on average, the first cut happened on the day of the year (DOY) 144 (24 May 2021), and that, except for one outlier, all parcels ($n=248$) were mown by DOY 180 (30 June 2021). However, the Andreatta et al. (2022) method estimated that 17% of the area was mown later than DOY 180. A detailed investigation revealed that this error was attributable to first cut omission errors, which meant that in those cases the first cut that was reported as having occurred was actually the second. Pixels with unrealistic first cut dates (i.e. >DOY 180) were therefore removed from the spatial averaging of mowing metrics (details about spatial averaging in Supporting Information 1.3). This adjusted method demonstrated very high accuracy of the estimated first cut dates (details in Supporting Information 1.4; Griffiths et al., 2020; Schwieder et al., 2022).

For the 73 point counts with a forage crop cover >0% within the bird sampling buffer (150 m), and for each spatial scale between 150 and 3000 m radius (i.e. $n=58$), we estimated the mean first forage cut date, the forage mowing frequency (i.e. the average mean number of forage cuts between April and mid-November), and the first forage cut date heterogeneity. The latter was expressed as Shannon's diversity of the first forage cut date at each point count and each scale accounting for the overall surface mown on the same date. Lower values of this metric suggest synchronous mowing and higher values asynchronous mowing.

2.5 | Scale-optimized effects of landscape composition/configuration on skylark abundance

We first modelled skylark abundance as a function of landscape composition/configuration. This was done on the full dataset (i.e. including plots without foraging crops within 150 m, $N=120$). We first applied the (pseudo-)optimized multiple scales method proposed by McGarigal et al. (2016). Specifically, we tested the effect of each predictor by univariate regression models fitted separately for each of the 58 scales considered; then, we selected the single best scale for each covariate, choosing the one showing the lowest AICc. A univariate approach was required to avoid issues related to the inherent correlation of landscape compositional variables. Subsequently, we built a multivariate (and multi-scale) model, including each predictor at the best-supported scale according to the previous step. This analysis was performed using GLMMs with a generalized Poisson distribution and a log link function (exploratory Poisson models indicated underdispersion). Continuous variables were standardized before model fitting. Following graphic exploratory analyses (Zuur et al., 2010), built-up area cover was $\log_{10}$-transformed to reduce the influence of outliers and reduce skewness. Point count cluster identity was entered in the model as a random intercept to account for potential non-independence of surveys related to the day of sampling and the spatial arrangement of the point counts (the 12 points in a cluster were relatively close and thus were not considered spatially independent). Multicollinearity was assessed with the adjusted generalized variance inflation factor (aGVIF) calculated with the R package *misty* (Yanagida, 2021) on the full model. When collinearity was considered too high (aGVIF >2.5; Johnston et al., 2018), we progressively removed the most collinear variable(s) starting from the one with the highest aGVIF and stopping when the metric fell below the threshold of 2.5. Forage, winter, and summer crop cover were tested with their quadratic term, as there is evidence (Assandri et al., 2023), supported by exploratory analyses, that ground-nesting species can show non-linear responses to these variables.

GLMMs were fitted in the R package *glmmTMB* (Brooks et al., 2017). Model validation was performed with the *DHARMa* R package (Hartig, 2021); no major model fit issues were detected in any of the final models. All the analyses were performed with R 4.2.2 (R Core Team, 2022).

2.6 | Scale-optimized effects of forage crop mowing on skylark abundance

We applied the (pseudo-) optimized multiple scales method described in Section 2.5 to the three forage mowing variables. Subsequently, we built a multivariate model, including each predictor at the resulting best-supported scale and, to control for the influence of landscape variables, we also included composition and configurational covariates at the best scale resulting from the previous step (Section 2.5). The analytical framework was the same as the one described in Section 2.5. In this model, the only plot in which four territories of skylarks were found was shown to be an influential point according to exploratory analysis and was thus removed from the analyses (Ieno & Zuur, 2015).

2.7 | Timing and consistency of forage crop mowing across years

To assess whether the timing of the first forage crop cut was consistent among years (i.e., if fields mown earlier in 2021 were consistently also mown earlier in previous years), we first tested whether this variable measured in 2021 was predicted by the same variable measured the year before and 3 years before (which represents the average skylark lifespan; Delius, 1965) using Gaussian GLMMs (with cluster identity as a random intercept). Additionally, we assessed the repeatability of the (standardized) first forage cut date over the 3 years using the R package *rptR* (Stoffel et al., 2017).

3 | RESULTS

3.1 | Scale-optimized effects of landscape composition/configuration

We detected at least one skylark territory in 55% of point counts (median = 1; range: 0–4 territories). The average number of skylark territories did not significantly vary among the three count sessions (χ^2 (df = 2) = 0.80; $p = 0.67$; Figure S2), suggesting that three visits were more than adequate.

The scale optimisation process for seven landscape predictors, each tested at 58 spatial scales (Figure 2), showed that the best scale for predicting the effect of forage crop cover on skylark abundance was 200 m, with a comparable level of support between 150 and 400 m, whereas for winter crop cover, it was 2600 m (1700 and 3000 m). The best, and only supported, scale for seminatural

FIGURE 2 Scale optimisation of landscape predictors. AICc values were derived from univariate regressions testing the effect of a given covariate on skylark abundance (models fitted for predictors calculated for each incremental 50 m buffer, 150–3000 m from the point count). The AICc difference (ΔAICc) from the best-fitting model is reported. Negative parameter estimates are reported in red, and positive ones in green. Transparency highlights non-significant associations ($p \geq 0.05$). Grey shadow represents models in an interval of ΔAIC = 2, having comparable support. $N = 120$ point counts.

habitat cover was 150 m, whereas for built-up areas, it was 200 m (with comparable support between 150 and 450 m). The best scale for AI was 350 m (150–500 m). For summer crop cover and SHDI, all the scales were comparably supported, and the null model was always the best-fitting one; thus, these variables were not included in the final model. In addition, we removed built-up areas$_{200}$, as it was moderately negatively correlated with AI$_{350}$ ($r = -0.42$), inflating the aGVIF metric.

According to the final scale-optimized multivariate landscape model (Table S1; Figure 3), skylark abundance increased with increasing extent of forage crops locally (200 m) and of winter crops at a wider scale (2600 m). Similarly, abundance increased where landscape patches were more aggregated (350 m) and decreased with increasing extent of seminatural habitats (150 m).

3.2 | Scale-optimized effects of forage crop mowing

The scale optimisation process for three forage crop mowing predictors, each tested at 58 different spatial scales (Figure 4), showed that the best scale for the effect of the first forage cut date was 1150 m, with a comparable level of support between 950 and 1850 m. For mowing frequency and forage crop cut date heterogeneity, all the scales were comparably supported and the null model was always the best-fitting one, thus these variables were not included in subsequent analyses. It is worth noting that first forage cut date and forage mowing frequency (at 1150 m) were moderately negatively correlated ($r = -0.43$; $p < 0.001$).

According to the final scale-optimized multivariate forage mowing model (Table S2; Figure 5), skylark abundance was higher when forage crops were mown later at a landscape scale (1150 m). This held true even when accounting for the scale-optimized landscape compositional and configurational covariates. In fact, the coefficients of the landscape covariates were comparable to those based on the 120-point model, taking into account the smaller sample size. The only difference was for forage crop cover$_{200}$, whose effect became non-significant because of the considerably lower variance in this subset of the data (CV$_{72}$ = 60%; CV$_{120}$ = 108%).

3.3 | Timing and consistency of first cut date across years

In 2021, the median DOY of the first cut was 142 (22.05), 135 (15.05) in 2020, and 146 (26.05) in 2018 (Figure S3). The mean first forage cut date in 2021 (at 1150 m) was significantly predicted by the same variable measured in the previous year ($\beta = 0.34$ [95% CI = 0.04–0.64]; χ^2 (df = 1) = 4.92; $p = 0.02$), and 3 years earlier ($\beta = 0.25$ [95% CI = 0.02–0.48]; χ^2 (df = 1) = 4.84; $p = 0.02$). Hence, the first cut date at the landscape scale was moderately, but significantly, repeatable across years ($R = 0.36$; $p = 0.001$), implying that a given farmland area experienced consistently early or late mowing.

4 | DISCUSSION

Our results showed that the local abundance of a declining farmland songbird, the skylark, responded to different habitat/management variables at specific spatial scales, which varied considerably among different predictors. Abundance increased with an increasing cover of forage crops locally and of winter crops at a landscape scale. In addition, skylarks were locally more abundant in aggregated crop patches. Thanks to our remote sensing approach, we showed that the timing of mowing affected the species at a landscape scale, with early-mown areas supporting fewer skylarks. Although several other studies have previously investigated the determinants of skylark abundance at the local versus landscape scale (Batáry et al., 2007; Chamberlain & Siriwardena, 2000; Miguet et al., 2013; Szilassi et al., 2022), they used fixed a priori-defined scales, and none consistently identified the exact spatial scale (or range of scales) at which different landscape variables affect skylark occurrence or abundance in agroecosystems. Additionally, to our knowledge, no previous study has explored the effect of mowing on skylark abundance beyond the local (i.e. crop parcel) scale.

Skylark abundance was enhanced by forage crops at the local scale (200 m), confirming that this habitat represents a grassland surrogate in intensive agroecosystems (Buckingham et al., 2015). Conversely, at a broader landscape scale (2600 m), the species showed a positive association with winter crop cover. Considering that, in our study area, these two crops are the dominant land cover, and that the higher the cover of winter crops, the lower those of forage crops, it can be concluded that skylarks locally prefer breeding in forage patches within a mixed farmland landscape. These results confirm the importance of landscape compositional heterogeneity for this species (Bretagnolle et al., 2019; Miguet et al., 2013; Püttmanns, Lehmann, et al., 2022), but also add a new element, that is a strong scale-specificity in the importance of the effects of different habitats on abundance of breeding pairs. In the study area, the coexistence of forage, winter, and summer crops (with the latter being less influential) is due to crop rotation, which is required for the production of forage for the local dairy industry and has prevented the establishment of monocultures (Assandri et al., 2023; Tabacco et al., 2018). Forage and winter crops have different phenologies, which result in different vegetation heights and structures in different periods of the breeding season. Although forage crops are likely the primary breeding habitat for the species, winter crops may constitute a complementary resource, possibly used for foraging or breeding when the forage crop sward height is inadequate, or for laying replacement clutches when forage crops are mown.

Considering landscape configuration, skylark abundance in the study area was positively associated with aggregated habitat patches at 350 m, in line with previous results showing how this open-habitat specialist favours larger fields (Batáry et al., 2007; Gayer et al., 2019). Although removed from the analysis due to collinearity, it is worth mentioning that built-up areas$_{200}$ were negatively associated with the aggregation index; this is best interpreted as an effect

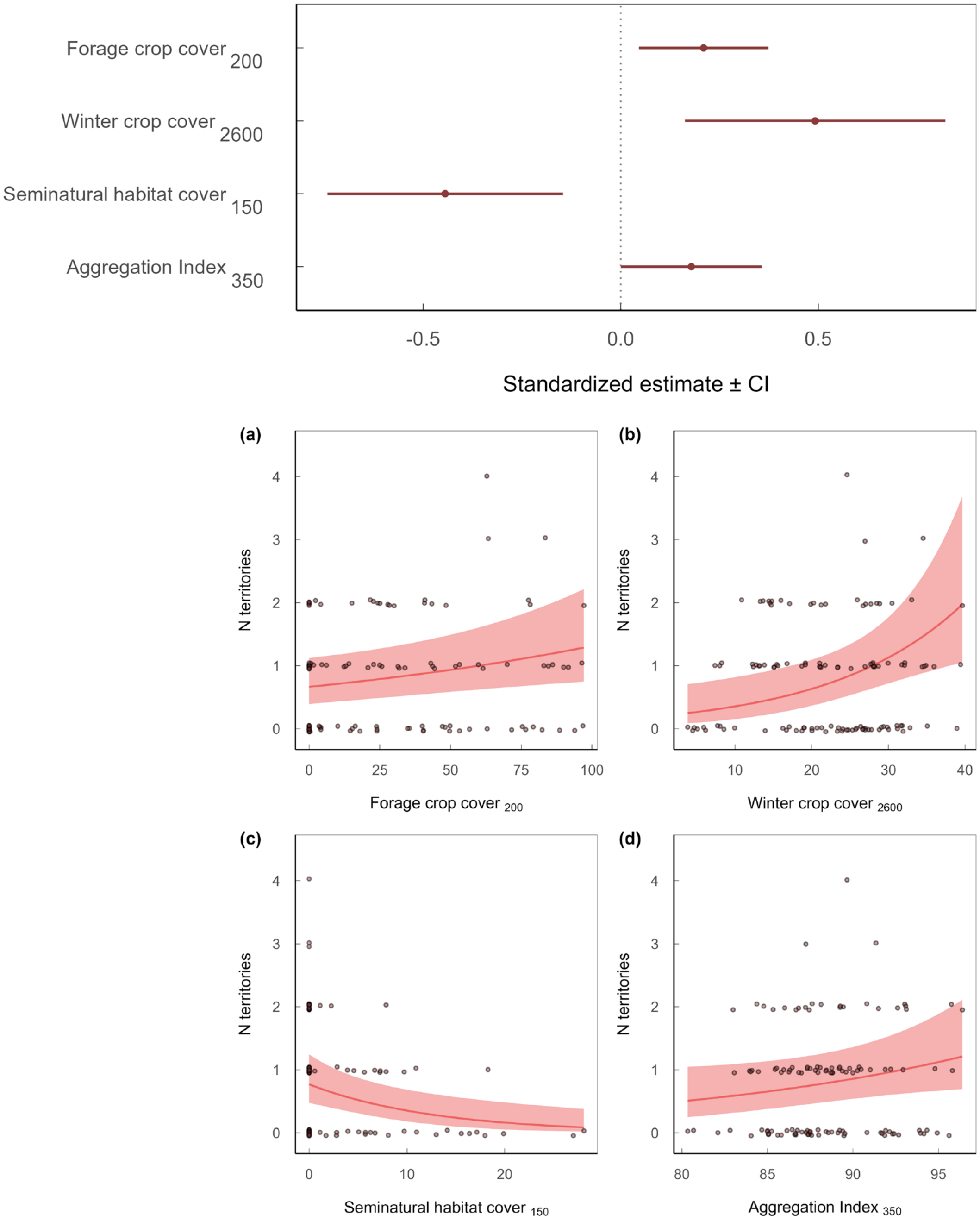

FIGURE 3 Top panel. Caterpillar plot showing the parameter estimates from a generalized Poisson GLMM of skylark abundance (number of territories) in relation to landscape covariates (Table S1). The spatial scale at which landscape predictors were entered in the models (i.e. the most supported according to the scale optimisation) is reported as subscripts. Bottom panel. Effects of landscape covariates ((a) forage crop cover; b) winter crop cover; c) seminatural habitat cover; d) aggregation index) on skylark abundance according to the same model. Only variables for which confidence intervals of estimates did not include zero are shown. Dots represent observed values. A small amount of jittering was applied to reduce overlap. Regression lines and 95% confidence bands are derived from the model and are shown in shaded colours. The scale at which each variable was entered in the model after scale optimisation is reported on the x-axis. Other predictors included in the model are kept at their mean values. *N* = 120 point counts.

of the agroecosystem fragmentation produced by buildings, roads and other infrastructures, which reduces habitat availability, results in higher human disturbance and increases nest predation (Assandri et al., 2017; Filippi-Codaccioni et al., 2008; Loretto et al., 2019). The same is likely true for the negative local-scale effect of seminatural habitats (e.g. small wetlands, isolated woodlots, tree rows, hedgerows), which are generally avoided by the species and other open-habitat specialists because they fragment the open landscape and increase predation risk (Bazzi et al., 2015; Canonne et al., 2024; Morris & Gilroy, 2008).

It could be expected that the date of the first cut of forage crops would have a greater effect on skylark abundance at a small

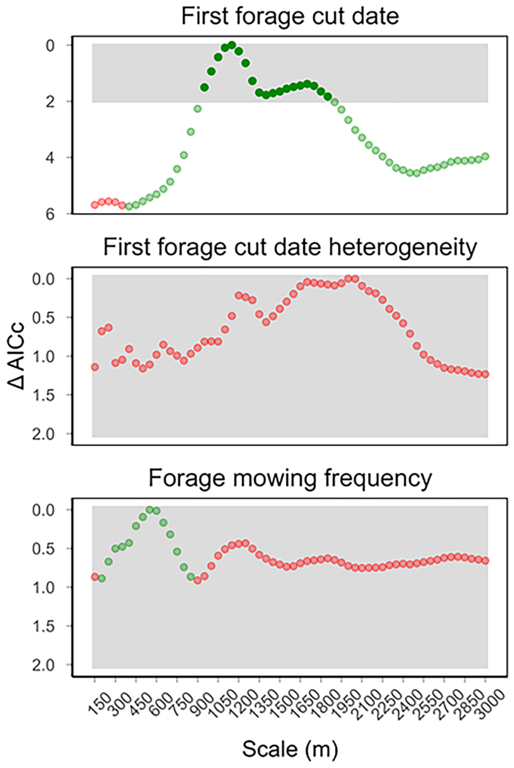

FIGURE 4 Scale optimisation of forage mowing predictors. AICc values were derived from univariate regressions testing the effect of a given covariate on skylark abundance (models fitted for predictors calculated for each incremental 50 m buffer, 150–3000 m from the point count). The AICc difference from the best-fitting model is reported. Negative parameter estimates are reported in red, and positive ones in green. Transparency highlights non-significant associations ($p \geq 0.05$). Grey shadow represents models in an interval of ΔAIC = 2, having comparable support. $N = 72$ points.

spatial scale (e.g. field) than at a broader scale. Indeed, forage crop mowing has been considered responsible for considerable nest losses, which, in the long run, may trigger population decline by reducing breeding productivity (Buckingham et al., 2015; Kragten et al., 2008; Stein-Bachinger & Fuchs, 2012; Wilson et al., 1997). Contrary to expectation, we detected a stronger effect at a broader landscape scale (950–1850 m; most supported 1150 m), with areas mown on average earlier supporting lower numbers of skylarks. This finding is robust as it simultaneously accounts for compositional and configuration landscape patterns. Although skylarks suffer a direct impact of mowing at the field scale (through, e.g. nest destruction and even adult mortality), our approach did not highlight such a pattern, as the mean number of territories in the third count session, after which most of the forage crops had been mown, was similar to previous counts (Figure S4). On the contrary, our novel remote sensing approach to estimate the timing of crop mowing highlighted a previously undocumented broad-scale spatial pattern; we showed that among years, at a scale broader than that of the field (which we identified to be approximately 1 km), the mowing calendar was quite consistent, that is areas mown on average earlier in a given year were also likely to be mown earlier in other years. Notably, those areas which were mown consistently earlier were those where skylark abundance was lowest. Hence, in areas constantly mown earlier, possibly due to pedological and climatic reasons, and, partly, to farm-level choices by farmers, the direct impact of mowing on skylark breeding success may be higher and this, in the long run, may turn early-mown forage crop patches into ecological traps with limited or null productivity, depressing local population size. Low skylark breeding success would be expected to promote breeding dispersal (ca. 1 km in skylark; Paradis et al., 1998). Breeding dispersal away from such areas may also depress local population size. Whether the consequent redistribution of breeding adults across the landscape, both within and between years, explains the observed patterns remains to be investigated.

Under our approach, we were able to identify which is the most significant effect of mowing on the species, that is the date of the first forage cut of the year. Other mowing variables, such as the intensity of mowing or the heterogeneity of the first cut date (which hypothetically could create a patchy landscape in which some nests or juveniles could survive), had no effect. We suggest that this is because the first forage cut is likely to occur at the end of the nestling rearing period of the first clutch (see Ferlini (2006) for a study in a comparable area), and its timing is thus crucial to breeding success. Considering that the impact of mowing on grassland birds is often context-specific (Canonne et al., 2024), our results should be considered valid for the study area and their replicability elsewhere should be evaluated on a case-by-case basis. Nevertheless, we believe that our approach based on scale optimisation and mowing covariates derived through remote sensing is promising and possibly applicable to other species/contexts.

Our novel findings emerged thanks to the state-of-the-art remote sensing approach proposed by Andreatta et al. (2022) and further developed here, reaching a very high accuracy compared with current mowing detection algorithms (Griffiths et al., 2020; Schwieder et al., 2022). This allowed us to accurately map mowing history at a landscape scale across years for the first time. Other approaches have previously aimed to estimate the effect of grassland mowing regimes on bird population occurrence/abundance. For instance, Howison et al. (2018) assessed the habitat preferences of black-tailed godwits (*Limosa limosa*) using as a proxy the temporal stability of vegetation types, which was calculated as the standard deviation of the variations in surface roughness obtained from synthetic aperture radar. Meanwhile, Bekkema and Eleveld (2018) examined the relationship between the distribution of nests in grasslands and a land use intensity index based on Sentinel-2 imagery. The explicit estimation of the first cut date and mowing frequency—the two specific metrics that are expected to have an impact on bird communities—together with the use of a cloud platform providing both access to images and huge computational capacity, are what make our approach novel. It gives the possibility of rapidly and efficiently mapping the date of the first cut in multiple years. This allowed us to show how the median date of the first cut can change considerably over the years, for

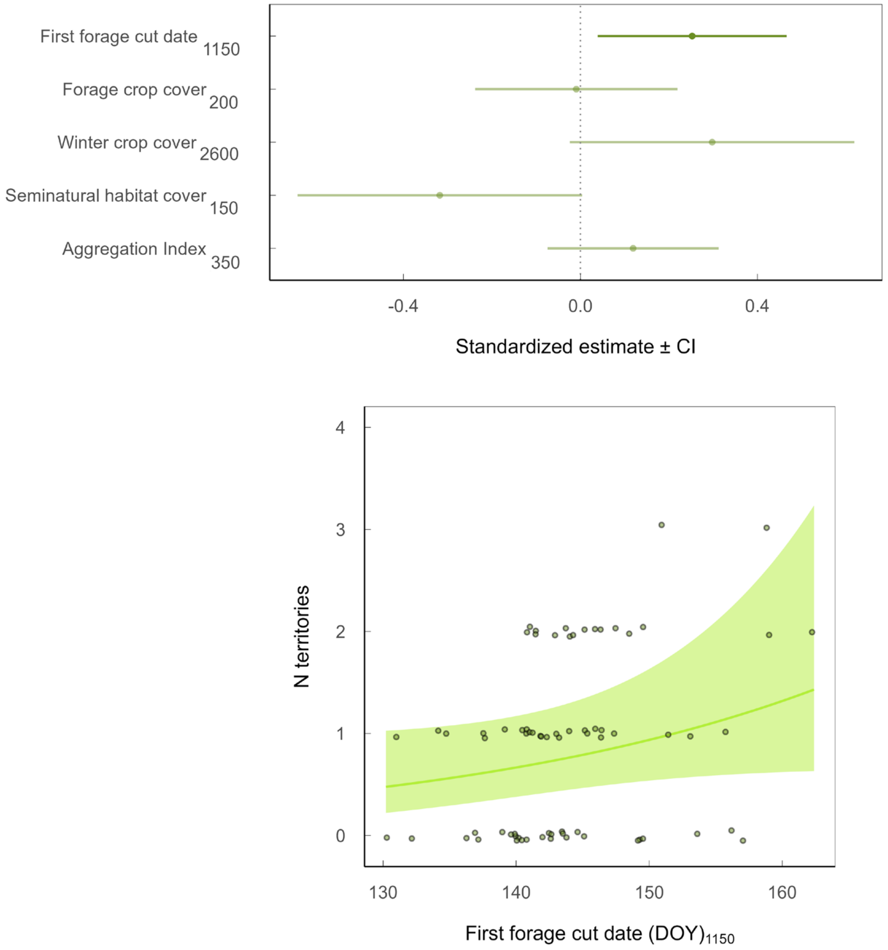

FIGURE 5 Top panel. Caterpillar plot showing parameter estimates from a generalized Poisson GLMM of skylark abundance (number of territories) in relation to first forage cut date (only supported forage crop mowing predictor) at its best-supported spatial scale (1150 m; Table S2), controlling for the landscape covariates at the best scale as resulting from landscape composition/configuration analysis (see Figure 3). The spatial scale at which landscape predictors were entered in the models (i.e. the most supported according to the scale optimisation) are reported as subscripts. The parameters whose confidence intervals encompass zero are shown in a lighter colour. Bottom panel. Effects of first forage cut date on skylark abundance according to the same model. Only variables for which confidence intervals of estimates did not include zero are shown. Dots represent observed values. A small amount of jittering was applied to reduce overlap. The regression line and 95% confidence bands are predicted from the model and are shown in shaded colours. Other predictors included in the model are kept at their mean values. $N=72$ point counts.

instance in 2020 it occurred on average 6–9 days before 2018 and 2021 (Figure S3), although the mowing calendar is consistently early or late across areas. This date depends on the climatic conditions of the year (mowing is delayed in cold and/or wet years) and might lead to high heterogeneity in skylark breeding success among years. However, the long-term effect of early mowing is likely reflected in the low abundance of the species in those areas, as shown in this study.

4.1 | Management and conservation implications

The agroecosystem of the southeastern Po Plain still harbours good numbers of skylarks, at least in the context of the alarming decline of the species in Emilia-Romagna, Italy, and across Europe (Brichetti & Fracasso, 2020; Keller et al., 2020; Rete Rurale Nazionale & Lipu, 2024a, 2024b; Tirozzi et al., 2024). In fact, the production of *Parmigiano Reggiano* still sustains a considerable extent of forage crops, particularly compared with the rest of the Po Plain, where maize (an unfavourable habitat for the species; Praus & Weidinger, 2015) covers more than 90% of the agricultural area (Bava et al., 2014; Tabacco et al., 2018). However, this area has been immune to the strong intensification experienced by European farmland (i.e. increase in farm size and consequent mechanization, reduction in habitat heterogeneity, and simplification of the agricultural landscape; see Calvi et al. (2018) and Assandri et al. (2023) for details on the study area) and the associated impacts on biodiversity (Assandri, 2022; Stoate et al., 2009). Urgent conservation measures and biodiversity-tailored landscape planning should thus be undertaken.

Our results can support the design of landscape scale-effective AESs using state-of-the-art remote sensing approaches, which allow the tracking of agricultural practices, such as mowing over large areas on an annual basis. This could be framed within the context of the new CAP reform, which started in 2023, and potentially provides 'green' instruments for halting the farmland biodiversity crisis. Among the measures defined under Pillar I, the new CAP has introduced eco-schemes, which should support farmers in adopting practices that minimize the negative impact of agriculture on the environment and climate (Guyomard et al., 2023). Based on our results, we suggest that crop rotation which includes forage and winter crops could be a good candidate for an eco-scheme, given its limited impact on farmers' income and strong beneficial outcomes on the environment and biodiversity. On the contrary, Pillar II agri-environmental and climatic measures, which pay farmers to apply biodiversity-friendly practices, can sustain more demanding (and beneficial) practices for the skylark and other ground-nesting species, primarily through the delayed mowing of forage crops. The spatial scale is crucial when designing agri-environment measures for this—or other ecologically similar—species given that it is evident that they are often applied at an inappropriate spatial scale (Sharps et al., 2023; Siriwardena, 2010; Walker et al., 2018). For instance, in the study area, the scale of application of AES is typically the farm, which has an average size of 14.5 ha (Calvi et al., 2018), that is less than 5% of the area in which the effect of the first forage cut is most likely to impact the species. Therefore, extending the spatial scale of interventions may significantly enhance the ecological success of a measure of this kind. This can be possibly achieved by gathering farmers in collectives, as already occurs in other parts of Europe (Grondard et al., 2023), which would allow a coordinated implementation of AESs at the scale of landscape rather than the individual farm.

Further research should focus on understanding the mechanism by which mowing activity affects the species, ultimately influencing its fitness, dispersal, and abundance. This could be achieved by leveraging population models, monitoring individual movements through miniaturized GPS devices (Wild et al., 2022), or by mark–recapture studies (Pérez-Granados et al., 2022). Additionally, regular monitoring of the species to address changes in its population size is needed to better understand the long-term effects of management practices and potential conservation measures.

AUTHOR CONTRIBUTIONS

Giacomo Assandri conceived the study and designed the methodology. Giacomo Assandri, Leonardo Siddi, Gaia Bazzi, and Riccardo Nardelli carried out fieldwork. Davide Andreatta and Giacomo Assandri led the analyses to which all authors contributed. Davide Andreatta and Giacomo Assandri led the writing of the manuscript with the help of Michelangelo Morganti, Diego Rubolini, Jacopo G. Cecere, and Dan Chamberlain. All authors contributed critically to the draft and gave final approval for publication.

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CONFLICT OF INTEREST STATEMENT

The authors declare no conflict of interest.

DATA AVAILABILITY STATEMENT

Data are available via the Figshare repository: https://doi.org/10.6084/m9.figshare.28020830.v1 (Assandri, 2024).

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REFERENCES

Andreatta, D., Gianelle, D., Scotton, M., Vescovo, L., & Dalponte, M. (2022). Detection of grassland mowing frequency using time series of vegetation indices from Sentinel-2 imagery. *GIScience & Remote Sensing*, *59*(1), 481–500. https://doi.org/10.1080/15481603.2022.2036055

Assandri, G. (2022). Editorial—Do ornithologists still play a role in reversing the crisis of farmland biodiversity? *Avocetta*, *46*, 3–5. https://doi.org/10.30456/AVO.2022E1

Assandri, G., Bazzi, G., Siddi, L., Nardelli, R., Cecere, J. G., Rubolini, D., & Morganti, M. (2023). The occurrence of a flagship raptor species in intensive agroecosystems is associated with more diverse farmland bird communities: Opportunities for market-based conservation. *Agriculture, Ecosystems & Environment*, *349*, 108441. https://doi.org/10.1016/j.agee.2023.108441

Assandri, G., Bernardi, A., Schmoliner, A., Bogliani, G., Pedrini, P., & Brambilla, M. (2018). A matter of pipes: Wryneck *Jynx torquilla* habitat selection and breeding performance in an intensive agroecosystem. *Journal of Ornithology*, *159*, 103–114. https://doi.org/10.1007/s10336-017-1479-y

Assandri, G., Bogliani, G., Pedrini, P., & Brambilla, M. (2017). Land-use and bird occurrence at the urban margins in the Italian Alps: Implications for planning and conservation. *North-Western Journal of Zoology*, *13*(1), 77–84.

Assandri, G., Bogliani, G., Pedrini, P., & Brambilla, M. (2019a). Toward the next Common Agricultural Policy reform: Determinants of avian communities in hay meadows reveal current policy's inadequacy for biodiversity conservation in grassland ecosystems. *Journal of Applied Ecology*, *56*(3), 604–617. https://doi.org/10.1111/1365-2664.13332

Assandri, G., Bogliani, G., Pedrini, P., & Brambilla, M. (2019b). Species-specific responses to habitat and livestock management call for carefully targeted conservation strategies for declining meadow birds. *Journal for Nature Conservation*, *52*, 125757. https://doi.org/10.1016/j.jnc.2019.125757

Assandri, G. (2024). Remote sensing reveals scale-specific effects of forage crop mowing and landscape structure on a declining farmland bird. *Figshare*. Dataset. https://doi.org/10.6084/m9.figshare.28020830.v1

Batáry, P., Báldi, A., & Erdős, S. (2007). Grassland versus non-grassland bird abundance and diversity in managed grasslands: Local, landscape and regional scale effects. *Biodiversity and Conservation*, *16*(4), 871–881. https://doi.org/10.1007/s10531-006-9135-5

Batáry, P., Dicks, L. V., Kleijn, D., & Sutherland, W. J. (2015). The role of agri-environment schemes in conservation and environmental management. *Conservation Biology: The Journal of the Society for Conservation Biology*, *29*(4), 1006–1016. https://doi.org/10.1111/cobi.12536

Battini, F., Agostini, A., Tabaglio, V., & Amaducci, S. (2016). Environmental impacts of different dairy farming systems in the Po Valley. *Journal of Cleaner Production*, *112*, 91–102. https://doi.org/10.1016/j.jclepro.2015.09.062

Bava, L., Sandrucci, A., Zucali, M., Guerci, M., & Tamburini, A. (2014). How can farming intensification affect the environmental impact of milk production? *Journal of Dairy Science*, *97*(7), 4579–4593. https://doi.org/10.3168/jds.2013-7530

Bazzi, G., Foglini, C., Brambilla, M., Saino, N., & Rubolini, D. (2015). Habitat management effects on Prealpine grassland bird communities. *The Italian Journal of Zoology*, *82*(2), 251–261. https://doi.org/10.1080/11250003.2014.983566

Bekkema, M. E., & Eleveld, M. (2018). Mapping grassland management intensity using Sentinel-2 satellite data mapping grassland management intensity using Sentinel-2 satellite data introduction actual grassland-use intensity maps are highly desirable for the study and conservation of. *GI_Forum*, *1*, 194–213. https://doi.org/10.1553/giscience2018

Bibby, C. J. (2000). *Bird census techniques*. Elsevier.

BirdLife International. (2017). *European birds of conservation concern: Populations, trends and national responsibilities*. Cambridge, UK: BirdLife International, 170.

Brambilla, M., Gubert, F., & Pedrini, P. (2021). The effects of farming intensification on an iconic grassland bird species, or why mountain refuges no longer work for farmland biodiversity. *Agriculture, Ecosystems and Environment*, *319*(May), 107518. https://doi.org/10.1016/j.agee.2021.107518

Bretagnolle, V., Siriwardena, G., Miguet, P., Henckel, L., & Kleijn, D. (2019). Local and landscape scale effects of heterogeneity in shaping bird communities and population dynamics. In *Agroecosystem diversity* (pp. 231–243). Elsevier. https://doi.org/10.1016/B978-0-12-811050-8.00014-5

Brichetti, P., & Fracasso, G. (2020). *Birds of Italy-2. Pteroclidae—Locustellidae*. Edizioni Belvedere.

Brooks, M. E., Kristensen, K., van Benthem, K. J., Magnusson, A., Berg, C. W., Nielsen, A., Skaug, H. J., Mächler, M., & Bolker, B. M. (2017). glmmTMB balances speed and flexibility among packages for zero-inflated generalized linear mixed modeling. *The R Journal*, *9*(2), 378. https://doi.org/10.32614/RJ-2017-066

Broyer, J., Curtet, L., & Boissenin, M. (2012). Does breeding success lead meadow passerines to select late mown fields? *Journal of Ornithology*, *153*(3), 817–823. https://doi.org/10.1007/s10336-011-0799-6

Buckingham, D. L., Giovannini, P., & Peach, W. J. (2015). Manipulating grass silage management to boost reproductive output of a ground-nesting farmland bird. *Agriculture, Ecosystems & Environment*, *208*, 21–28. https://doi.org/10.1016/j.agee.2015.04.018

Burfield, I. J., Rutherford, C. A., Fernando, E., Grice, H., Piggott, A., Martin, R. W., Balman, M., Evans, M. I., & Staneva, A. (2023). Birds in Europe 4: The fourth assessment of species of European conservation concern. *Bird Conservation International*, *33*, e66. https://doi.org/10.1017/S0959270923000187

Calvi, G., Campedelli, T., Tellini Florenzano, G., & Rossi, P. (2018). Evaluating the benefits of agri-environment schemes on farmland bird communities through a common species monitoring programme. A case study in northern Italy. *Agricultural Systems*, *160*(August 2016), 60–69. https://doi.org/10.1016/j.agsy.2017.09.002

Canonne, C., Chiffard, J., Curtet, L., & Besnard, A. (2024). Response of grassland birds to local features strongly depends on landscape context. *Agriculture, Ecosystems & Environment*, *365*(February), 108905. https://doi.org/10.1016/j.agee.2024.108905

Chamberlain, D. E., & Siriwardena, G. M. (2000). The effects of agricultural intensification on skylarks (Alauda arvensi): Evidence from monitoring studies in Great Britain. *Environmental Reviews*, *8*(2), 95–113. https://doi.org/10.1139/a00-007

Copland, A. S., Crowe, O., Wilson, M. W., & O'Halloran, J. (2012). Habitat associations of Eurasian Skylarks Alauda arvensis breeding on Irish farmland and implications for agri-environment planning. *Bird Study*, *59*(2), 155–165. https://doi.org/10.1080/00063657.2011.652593

Delius, J. D. (1965). A population study on skylarks Alauda arvensis. *Ibis*, *107*, 466–492.

Donald, P. F., & Harris, A. (2010). *The skylark*. T & A D Poyser.

Douglas, D. J. T., Waldinger, J., Buckmire, Z., Gibb, K., Medina, J. P., Sutcliffe, L., Beckmann, C., Collar, N. J., Jansen, R., Kamp, J., Little, I., Sheldon, R., Yanosky, A., & Koper, N. (2023). A global review identifies agriculture as the main threat to declining grassland birds. *Ibis*, *165*, 1107–1128. https://doi.org/10.1111/IBI.13223

Falcucci, A., Maiorano, L., & Boitani, L. (2007). Changes in land-use/land-cover patterns in Italy and their implications for biodiversity conservation. *Landscape Ecology*, *22*(4), 617–631. https://doi.org/10.1007/s10980-006-9056-4

Ferlini, F. (2006). Biologia dell'Allodola, Alauda arvensis, in periodo riproduttivo in un'area agricola lombarda. *Rivista Italiana Di Ornitologia*, *76*(November 2007), 131–138.

Filippi-Codaccioni, O., Devictor, V., Clobert, J., & Julliard, R. (2008). Effects of age and intensity of urbanization on farmland bird communities. *Biological Conservation*, *141*(11), 2698–2707. https://doi.org/10.1016/j.biocon.2008.08.006

Gayer, C., Kurucz, K., Fischer, C., Tscharntke, T., & Batáry, P. (2019). Agricultural intensification at local and landscape scales impairs farmland birds, but not skylarks (*Alauda arvensis*). *Agriculture, Ecosystems & Environment*, *277*, 21–24. https://doi.org/10.1016/J.AGEE.2019.03.006

Green, R. E. (1996). Factors affecting the population density of the corncrake *Crex crex* in Britain and Ireland. *Journal of Applied Ecology*, *33*(2), 237–248.

Gregory, R. D., Eaton, M. A., Burfield, I. J., Grice, P. V., Howard, C., Klvaňová, A., Noble, D., Šilarová, E., Staneva, A., Stephens, P. A., Willis, S. G., Woodward, I. D., & Burns, F. (2023). Drivers of the changing abundance of European birds at two spatial scales. *Philosophical Transactions of the Royal Society, B: Biological Sciences*, *378*(1881), 20220198. https://doi.org/10.1098/rstb.2022.0198

Griffiths, P., Nendel, C., Pickert, J., & Hostert, P. (2020). Towards national-scale characterization of grassland use intensity from integrated Sentinel-2 and Landsat time series. *Remote Sensing of Environment*, *238*, 111124. https://doi.org/10.1016/j.rse.2019.03.017

Grondard, N., Kleyheeg, E., Hein, L., & Van Bussel, L. G. J. (2023). Effects of Dutch agri-environmental field margins and bird plots on cropland birds. *Agriculture, Ecosystems & Environment*, *349*, 108430. https://doi.org/10.1016/J.AGEE.2023.108430

Guerrero, I., Morales, M. B., Oñate, J. J., Geiger, F., Berendse, F., Snoo, G. d., Eggers, S., Pärt, T., Bengtsson, J., Clement, L. W., Weisser, W. W., Olszewski, A., Ceryngier, P., Hawro, V., Liira, J., Aavik, T., Fischer, C., Flohre, A., Thies, C., & Tscharntke, T. (2012). Response of ground-nesting farmland birds to agricultural intensification across Europe: Landscape and field level management factors. *Biological Conservation*, *152*, 74–80. https://doi.org/10.1016/J.BIOCON.2012.04.001

Guttery, M. R., Ribic, C. A., Sample, D. W., Paulios, A., Trosen, C., Dadisman, J., Schneider, D., & Horton, J. A. (2017). Scale-specific habitat relationships influence patch occupancy: Defining neighborhoods to optimize the effectiveness of landscape-scale grassland bird conservation. *Landscape Ecology*, *32*(3), 515–529. https://doi.org/10.1007/s10980-016-0462-y

Guyomard, H., Détang-Dessendre, C., Dupraz, P., Delaby, L., Huyghe, C., Peyraud, J. L., Reboud, X., & Sirami, C. (2023). How the Green architecture of the 2023–2027 common agricultural policy could have been greener. *Ambio*, *52*(8), 1327–1338. https://doi.org/10.1007/S13280-023-01861-0/FIGURES/2

Hartig, F. (2021). DHARMa: Residual diagnostics for hierarchical (multi-level/mixed) regression models. R package version 0.4.4. https://CRAN.R-project.org/package=DHARMa. https://cran.r-project.org/package=DHARMa

Hesselbarth, M. H. K., Sciaini, M., With, K. A., Wiegand, K., & Nowosad, J. (2019). landscapemetrics: An open-source R tool to calculate landscape metrics. *Ecography*, *42*(10), 1648–1657. https://doi.org/10.1111/ecog.04617

Howison, R. A., Piersma, T., Kentie, R., Hooijmeijer, J. C. E. W., & Olff, H. (2018). Quantifying landscape-level land-use intensity patterns through radar-based remote sensing. *Journal of Applied Ecology*, *55*(3), 1276–1287. https://doi.org/10.1111/1365-2664.13077

Ieno, E. N., & Zuur, A. F. (2015). *A Beginner's guide to data exploration and visualisation with R*. Highland Statistic Ltd.

Johnston, R., Jones, K., & Manley, D. (2018). Confounding and collinearity in regression analysis: A cautionary tale and an alternative procedure, illustrated by studies of British voting behaviour. *Quality & Quantity*, *52*(4), 1957–1976. https://doi.org/10.1007/s11135-017-0584-6

Keller, V., Herrando, S., Voříšek, P., Franch, M., Kipson, M., Milanesi, P., Martí, D., Anton, M., Klvanová, A., Kalyakin, M., & Bauer, H. (2020). *European breeding bird atlas 2: Distribution, abundance and change*. European Bird Census Council & Lynx Edition.

Kleijn, D., Baquero, R. A., Clough, Y., Díaz, M., De Esteban, J., Fernández, F., Gabriel, D., Herzog, F., Holzschuh, A., Jöhl, R., Knop, E., Kruess, A., Marshall, E. J. P., Steffan-Dewenter, I., Tscharntke, T., Verhulst, J., West, T. M., & Yela, J. L. (2006). Mixed biodiversity benefits of agri-environment schemes in five European countries. *Ecology Letters*, *9*(3), 243–257. https://doi.org/10.1111/j.1461-0248.2005.00869.x

Kragten, S., Trimbos, K. B., & de Snoo, G. R. (2008). Breeding skylarks (*Alauda arvensis*) on organic and conventional arable farms in The Netherlands. *Agriculture, Ecosystems & Environment*, *126*(3–4), 163–167. https://doi.org/10.1016/j.agee.2008.01.021

Lambin, E. F., Turner, B. L., Geist, H. J., Agbola, S. B., Angelsen, A., Bruce, J. W., Coomes, O. T., Dirzo, R., Fischer, G., Folke, C., George, P. S., Homewood, K., Imbernon, J., Leemans, R., Li, X., Moran, E. F., Mortimore, M., Ramakrishnan, P. S., Richards, J. F., … Xu, J. (2001). The causes of land-use and land-cover change: Moving beyond the myths. *Global Environmental Change*, *11*(4), 261–269. https://doi.org/10.1016/S0959-3780(01)00007-3

Li, Y., Piersma, T., Hooijmeijer, J. C. E. W., & Howison, R. A. (2023). Land-use intensity impacts habitat selection of ground-nesting farmland birds in The Netherlands. *Ecological Solutions and Evidence*, *4*(1), e12201. https://doi.org/10.1002/2688-8319.12201

Loretto, M.-C., Schöll, E. M., & Hille, S. (2019). Occurrence of Eurasian Skylark *Alauda arvensis* territories in relation to urban area and heterogeneous farmland. *Bird Study*, *66*(2), 273–278. https://doi.org/10.1080/00063657.2019.1637816

MacDonald, D., Crabtree, J., Wiesinger, G., Dax, T., Stamou, N., Fleury, P., Gutierrez Lazpita, J., & Gibon, A. (2000). Agricultural abandonment in mountain areas of Europe: Environmental consequences and policy response. *Journal of Environmental Management*, *59*(1), 47–69. https://doi.org/10.1006/jema.1999.0335

Mantovi, P., Dal Prà, A., Pacchioli, M. T., & Ligabue, M. (2015). Forage production and use in the dairy farming systems of Northern Italy. *Grassland Science in Europe-Gassland and Forages in High Output Dairy Farming Systems*, *20*(April), 67–77.

McCracken, M. E., Woodcock, B. A., Lobley, M., Pywell, R. F., Saratsi, E., Swetnam, R. D., Mortimer, S. R., Harris, S. J., Winter, M., Hinsley, S., & Bullock, J. M. (2015). Social and ecological drivers of success in agri-environment schemes: The roles of farmers and environmental context. *Journal of Applied Ecology*, *52*(3), 696–705. https://doi.org/10.1111/1365-2664.12412

McGarigal, K., Wan, H. Y., Zeller, K. A., Timm, B. C., & Cushman, S. A. (2016). Multi-scale habitat selection modeling: A review and outlook. *Landscape Ecology*, *31*(6), 1161–1175. https://doi.org/10.1007/s10980-016-0374-x

Miguet, P., Gaucherel, C., & Bretagnolle, V. (2013). Breeding habitat selection of Skylarks varies with crop heterogeneity, time and spatial scale, and reveals spatial and temporal crop complementation. *Ecological Modelling*, *266*(1), 10–18. https://doi.org/10.1016/j.ecolmodel.2013.06.029

Morris, A. J., & Gilroy, J. J. (2008). Close to the edge: Predation risks for two declining farmland passerines. *Ibis*, *150*(Suppl.1), 168–177. https://doi.org/10.1111/J.1474-919X.2008.00857.X

Paradis, E., Baillie, S. R., Sutherland, W. J., & Gregory, R. D. (1998). Patterns of natal and breeding dispersal in birds. *Journal of Animal Ecology*, *67*(4), 518–536. https://doi.org/10.1046/j.1365-2656.1998.00215.x

Pe'er, G., Dicks, L. V., Visconti, P., Arlettaz, R., Báldi, A., Benton, T. G., Collins, S., Dieterich, M., Gregory, R. D., Hartig, F., Henle, K., Hobson, P. R., Kleijn, D., Neumann, R. K., Robijns, T., Schmidt, J., Shwartz, A., Sutherland, W. J., Turbé, A., … Scott, A. V. (2014). EU agricultural reform fails on biodiversity. *Science*, *344*(6188), 1090–1092.

Pe'er, G., Zinngrebe, Y., Moreira, F., Sirami, C., Schindler, S., Müller, R., Bontzorlos, V., Clough, D., Bezák, P., Bonn, A., Hansjürgens, B., Lomba, A., Möckel, S., Passoni, G., Schleyer, C., Schmidt, J., & Lakner, S. (2019). A greener path for the EU common agricultural policy. *Science*, *365*(6452), 449–451. https://doi.org/10.1126/science.aax3146

Pelletier, N., & Tyedmers, P. (2010). Forecasting potential global environmental costs of livestock production 2000-2050. *Proceedings of the National Academy of Sciences of the United States of America*, *107*(43), 18371–18374. https://doi.org/10.1073/pnas.1004659107

Pérez-Granados, C., Sáez-Gómez, P., & López-Iborra, G. M. (2022). Breeding dispersal movements of Dupont's lark *Chersophilus duponti* in fragmented landscape. *Bird Conservation International*, *32*(1), 53–63. https://doi.org/10.1017/S095927092100006X

Pickett, S. R. A., & Siriwardena, G. M. (2011). The relationship between multi-scale habitat heterogeneity and farmland bird abundance. *Ecography*, *34*(6), 955–969. https://doi.org/10.1111/j.1600-0587.2011.06608.x

Praus, L., & Weidinger, K. (2015). Breeding biology of skylarks *Alauda arvensis* in maize and other crop fields. *Acta Ornithologica*, *50*(1), 59–68. https://doi.org/10.3161/00016454AO2015.50.1.007

Püttmanns, M., Böttges, L., Filla, T., Lehmann, F., Martens, A. S., Siegel, F., Sippel, A., Von Bassi, M., Balkenhol, N., Waltert, M., & Gottschalk, E. (2022). Habitat use and foraging parameters of breeding Skylarks indicate no seasonal decrease in food availability in heterogeneous farmland. *Ecology and Evolution*, *12*, 8461. https://doi.org/10.1002/ece3.8461

Püttmanns, M., Lehmann, F., Willert, F., Heinz, J., Kieburg, A., Filla, T., Balkenhol, N., Waltert, M., & Gottschalk, E. (2022). No seasonal curtailment of the Eurasian Skylark's (*Alauda arvensis*) breeding season in German heterogeneous farmland. *Ecology and Evolution*, *12*(9), e9267. https://doi.org/10.1002/ece3.9267

Pykälä, J. (2000). Mitigating human effects on European biodiversity through traditional animal husbandry. *Conservation Biology*, *14*(3), 705–712. https://doi.org/10.1046/j.1523-1739.2000.99119.x

R Core Team. (2022). *R: A language and environment for statistical computing*. R Foundation for Statistical Computing. http://www.r-project.org/

Rete Rurale Nazionale & Lipu. (2024a). Farmland Bird Index nazionale e andamenti di popolazione delle specie in Italia nel periodo 2000–2023.

Rete Rurale Nazionale & Lipu. (2024b). Emilia Romagna—Farmland Bird Index e andamenti di popolazione delle specie 2000-2023.

Schwieder, M., Wesemeyer, M., Frantz, D., Pfoch, K., Erasmi, S., Pickert, J., Nendel, C., & Hostert, P. (2022). Mapping grassland mowing events across Germany based on combined Sentinel-2 and Landsat 8 time series. *Remote Sensing of Environment*, *269*, 112795. https://doi.org/10.1016/j.rse.2021.112795

Sharps, E., Hawkes, R. W., Bladon, A. J., Buckingham, D. L., Border, J., Morris, A. J., Grice, P. V., & Peach, W. J. (2023). Reversing declines in farmland birds: How much agri-environment provision is needed at farm and landscape scales? *Journal of Applied Ecology*, *60*(4), 568–580. https://doi.org/10.1111/1365-2664.14338

Siriwardena, G. M. (2010). The importance of spatial and temporal scale for agri-environment scheme delivery. *Ibis*, *152*(3), 515–529. https://doi.org/10.1111/j.1474-919X.2010.01034.x

Stein-Bachinger, K., & Fuchs, S. (2012). Protection strategies for farmland birds in legume–grass leys as trade-offs between nature conservation and farmers' needs. *Organic Agriculture*, *2*(2), 145–162. https://doi.org/10.1007/s13165-012-0029-4

Stoate, C., Báldi, A., Beja, P., Boatman, N. D., Herzon, I., van Doorn, A., de Snoo, G. R., Rakosy, L., & Ramwell, C. (2009). Ecological impacts of early 21st century agricultural change in Europe—A review. *Journal of Environmental Management*, *91*(1), 22–46. https://doi.org/10.1016/J.JENVMAN.2009.07.005

Stoffel, M. A., Nakagawa, S., & Schielzeth, H. (2017). rptR: Repeatability estimation and variance decomposition by generalized linear mixed-effects models. *Methods in Ecology and Evolution*, *8*(11), 1639–1644. https://doi.org/10.1111/2041-210X.12797

Strebel, G., Jacot, A., Horch, P., & Spaar, R. (2015). Effects of grassland intensification on Whinchats *Saxicola rubetra* and implications for conservation in upland habitats. *Ibis*, *157*(2), 250–259. https://doi.org/10.1111/ibi.12250

Szilassi, P., Gallé, R., Szép, T., & Csikós, N. (2022). Scale dependence of landscape-structure-based estimation of abundance of Eurasian skylark (*Alauda arvensis*). *Ecological Indicators*, *139*, 108931. https://doi.org/10.1016/j.ecolind.2022.108931

Tabacco, E., Comino, L., & Borreani, G. (2018). Production efficiency, costs and environmental impacts of conventional and dynamic forage systems for dairy farms in Italy. *European Journal of Agronomy*, *99*(June), 1–12. https://doi.org/10.1016/j.eja.2018.06.004

Tirozzi, P., Orioli, V., Dondina, O., & Bani, L. (2024). Connection between ecological niche changes and population trends in the Eurasian Skylark (*Alauda arvensis*) breeding in lowland and mountain areas of southern Europe. *Ibis*, *166*, 1311–1328. https://doi.org/10.1111/ibi.13322

Veen, P., Jefferson, R., De Schmidt, J., & Van Der Straaten, J. (2009). *Grassland in Europe of high nature value*. KNNV Publishing.

Walker, L. K., Morris, A. J., Cristinacce, A., Dadam, D., Grice, P. V., & Peach, W. J. (2018). Effects of higher-tier agri-environment scheme on the abundance of priority farmland birds. *Animal Conservation*, *21*(3), 183–192. https://doi.org/10.1111/acv.12386

Wild, T. A., Koblitz, J. C., Dechmann, D. K. N., Dietz, C., Meboldt, M., & Wikelski, M. (2022). Micro-sized open-source and low-cost GPS loggers below 1 g minimise the impact on animals while collecting thousands of fixes. *PLoS One*, *17*(6), e0267730. https://doi.org/10.1371/journal.pone.0267730

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*, *34*(6), 1462–1478.

Yanagida, T. (2021). misty: Miscellaneous functions "T. Yanagida". R package version 0.4.3. https://CRAN.R-project.org/package=misty. https://cran.r-project.org/package=misty

Zuckerberg, B., Cohen, J. M., Nunes, L. A., Bernath-Plaisted, J., Clare, J. D. J., Gilbert, N. A., Kozidis, S. S., Maresh Nelson, S. B., Shipley, A. A., Thompson, K. L., & Desrochers, A. (2020). A review of overlapping landscapes: Pseudoreplication or a red herring in landscape ecology? *Current Landscape Ecology Reports*, *5*(4), 140–148. https://doi.org/10.1007/s40823-020-00059-4

Zuur, A. F., Ieno, E. N., & Elphick, C. S. (2010). A protocol for data exploration to avoid common statistical problems. *Methods in Ecology and Evolution*, *1*(1), 3–14. https://doi.org/10.1111/j.2041-210X.2009.00001.x

SUPPORTING INFORMATION

Additional supporting information can be found online in the Supporting Information section at the end of this article.

Figure S1. Example of Land Use/Land Cover (LULC) and one forage mowing parameter (i.e. first cut date) assessment at a point count.

Figure S2. Correlation plot of the first forage cut date expressed as DOY and obtained by two alternative methods, PlanetScope and Sentinel 2 (see text).

Figure S3. Distribution of the mean first forage cut date (1150 m scale), expressed as day of the year (DOY) in the study area in 2018, 2020, and 2021.

Figure S4. Number of skylark territories counted in the three field sessions.

Table S1. Results of a generalized Poisson GLMM testing the abundance of skylarks as a function of landscape covariates.

Table S2. Results of a generalized Poisson GLMM testing the abundance of skylarks as a function of the first forage cut date (only supported forage moving predictor) at its best supported spatial scale (1150 m) and controlling for the landscape covariates at best scale as resulting from landscape composition/configuration analysis.

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