



## The implementation of irrigation leads to declines in farmland birds

Xabier Cabodevilla<sup>a,b,c,\*</sup>, Alexander D. Wright<sup>c,d</sup>, Diego Villanua<sup>e</sup>, Beatriz Arroyo<sup>b,1</sup>,  
Elise F. Zipkin<sup>c,d,1</sup>

<sup>a</sup> Department of Zoology and Animal Cell Biology, Faculty of Pharmacy, University of the Basque Country (UPV/EHU), Paseo de la Universidad 7, 01006 Vitoria-Gasteiz, Alava, Spain

<sup>b</sup> Instituto de Investigación en Recursos Cinegéticos (IREC) (CSIC-UCLM-JCCM), Ronda de Toledo 12, 13005 Ciudad Real, Spain

<sup>c</sup> Department of Integrative Biology, Michigan State University, East Lansing, MI, USA

<sup>d</sup> Ecology, Evolution, and Behavior Program, Michigan State University, East Lansing, MI, USA

<sup>e</sup> Navarra Environmental Management (GAN-NIK), c/ Padre Adoain, 219, 31015 Pamplona, Navarra, Spain

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### ABSTRACT

Assessing the effects of agricultural intensification on biodiversity is critical for developing effective management plans for farmland conservation. Among other factors, the direct and indirect impacts of irrigation on wildlife have yet to be thoroughly studied despite significant increases in the surface area of irrigated farmlands since the mid-twentieth century (currently greater than 300 million hectares worldwide). Here, we evaluate the impact of irrigation on bird species occurrence patterns using a BACI (Before-After Control-Impact) design. Our study occurs in a 100 km<sup>2</sup> area with rainfed agriculture in the Mediterranean region of northern Spain. We analysed a 13-year dataset comprised of the 47 most common bird species in the region using a multi-species hierarchical occurrence model. We examined how the implementation of irrigation in a rain-fed farmland area altered the local bird community, identifying which species were negatively or positively impacted by changes to the local ecosystem. The implementation of irrigation had an overall negative impact on the bird community, with occurrence rates of most species (55%) decreasing and only a small fraction (11%) increasing after the onset of irrigation, leading to an overall reduction in site-level species richness. Irrigation had the most detrimental impact on farmland birds (including steppe birds, which are of high conservation concern), but also had negative effects on forest, shrubland, and non-specialist bird species that occur frequently in rainfed agricultural environments. The observed negative impacts on bird occurrences are likely due to the loss of nesting and foraging habitat arising from shifts in crops and/or loss of fallow lands associated with irrigation. The fact that only a few species responded positively to the implementation of irrigation suggests that in the long-term irrigation may lead to substantial negative changes in local bird communities, with less diversity and a lack of ecologically-important farmland species. Irrigation schemes should thus be implemented carefully, avoiding areas with high species richness or high densities of endangered species. In cases where irrigation cannot be avoided, promoting diverse agrosystems, avoiding monocultures, and including interspersed rainfed crops and fallow lands may help to mitigate negative effects on local bird communities and their ecosystems.

### 1. Introduction

Anthropogenic activities cause multiple impacts on natural and human-dominated landscapes, which can have direct and indirect effects on wildlife (Pimm et al., 1995; Hooke et al., 2012). Agriculture is widespread throughout the world and greatly changes habitat, nutrient loads and water availability, and thus, local and regional biodiversity

(Matson et al., 1997; Pain and Pienkowski, 1997; Benton et al., 2003). Within the last several decades, agriculture has intensified through increases in land conversion rates, average plot sizes, mechanization and use of high-yield crop varieties, irrigation, fertilizers, and pesticides (Matson et al., 1997). Together, these agricultural intensification measures can have detrimental effects on local and regional plant and animal population abundances and species richness (Pain and Pienkowski,

\* Corresponding author at: Department of Zoology and Animal Cell Biology, Faculty of Pharmacy, University of the Basque Country (UPV/EHU), Paseo de la Universidad 7, 01006 Vitoria-Gasteiz, Alava, Spain.

E-mail address: [xabier.cabodevilla@ehu.eus](mailto:xabier.cabodevilla@ehu.eus) (X. Cabodevilla).

<sup>1</sup> Shared last authorship

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1997; Stoate et al., 2001; Benton et al., 2002; Donald et al., 2006; Storkey et al., 2012).

The implementation and widespread use of irrigation (the artificial application of water to the soil to supplement direct rainfall) has been an important advancement for agricultural productivity (Alauddin and Quiggin, 2008), but also significantly alters local ecosystems (Baldock et al., 2000; Stoate et al., 2001). Irrigated farmland surface has increased dramatically in the last 50 years (Matson et al., 1997; Alauddin and Quiggin, 2008), currently covering more than 300 million hectares worldwide (FAO, 2011). This, combined with the increased use of fertilisers, has greatly boosted crop yields (Matson et al., 1997), facilitated the use of water-demanding crops in areas where they have been traditionally absent (Fagúndez et al., 2016; Giralt et al., 2021), and rendered the maintenance of fallows unnecessary (Baldock et al., 1994). For example, 1.4 million hectares of agricultural land have been converted to irrigation in Spain alone since 1973 (Baldock et al., 1994), and concurrently the proportion of arable land left fallow declined from > 50% in 1973 to < 40% in 1990 (Baldock et al., 1994), and has further declined by another 16–40% in just the last two decades (Traba and Morales, 2019). Similar trends have been documented in other Mediterranean countries like Greece, Italy, and Portugal (Caraveli, 2000). However, fallow farmland is an essential part of traditional rainfed (i.e. non-irrigated) cereal farming systems in Mediterranean biomes (Bignal and McCracken, 1996; Baldock et al., 2000; Caraveli, 2000) and provides critical habitat for many wildlife species in human-dominated landscapes (Denys and Tschardtke, 2002; Schmidt and Tschardtke, 2005; Vickery et al., 2004; Ng et al., 2017; Traba and Morales, 2019), including threatened steppe bird species (Sanderson et al., 2013; García de la Morena et al., 2018; Cabodevilla et al., 2020). Additionally, when fertilisers are distributed through irrigation schemes (fertigation), nitrate concentration can reach dangerous levels for wildlife drinking irrigated water (Rodríguez-Estival et al., 2010). Irrigation also influences soil properties through salinization and waterlogging (Matson et al., 1997; Baldock et al., 2000), which can further affect wildlife species distribution and abundance patterns. Cumulatively, these direct and indirect impacts suggest that irrigation leads to significant changes in land-use, vegetation structure and potentially local wildlife communities (Andrey et al., 2014; Fagúndez et al., 2016; Cabodevilla et al., 2021a, 2021b; Giralt et al., 2021).

The shift from rainfed farmland to irrigated farmland, including the changes to agrosystems associated with irrigation, can harm some species while benefiting others (Baraibar et al., 2009; González-Estébanez et al., 2011; Andrey et al., 2014; Pérez-Fuertes et al., 2015). Even for farmland birds, one of the groups most affected by agricultural intensification (Donald et al., 2001, 2006; Brennan and Kuvlesky, 2005), the effects of irrigation vary, with some species responding negatively and others adapting well. Steppe birds tend to avoid irrigated land (Brotons et al., 2004; De Frutos et al., 2015), while other species (e.g. harriers *Circus* spp and storks *Ciconia ciconia*) are known to use irrigated farmlands (Cardador et al., 2011; De Frutos et al., 2015; Torres-Orozco et al., 2016). As compared to rainfed farmland, irrigated lands can have large quantities of certain prey types (e.g. voles; Jareño et al., 2015), and/or denser vegetation, which may benefit some species for foraging or breeding (Cardador et al., 2011; Torres-Orozco et al., 2016). Although the effects of implementing irrigation within Mediterranean rainfed farmlands seem to vary among groups of species, most bird species are likely to be negatively impacted by irrigation because of the many associated changes in the landscape. However, the limited research on bird responses to irrigation has mostly focused on a few individual species; analyses on the broader community-level effects of the implementation of irrigation are scarce (but see Giralt et al., 2021). Such information is critical to developing conservation management plans for bird communities, especially as the implementation of irrigation is projected to continue increasing (Matson et al., 1997; Alauddin and Quiggin, 2008; FAO, 2011).

Here, we estimate the impact of the implementation of irrigation on a

community of bird species in a rain-fed farmland system using a BACI (Before-After Control-Impact; Popescu et al., 2012) design. We used a 13-year dataset to examine the effects of implementing irrigation on the species- and community-level occurrence patterns of 47 of the most common bird species in a Mediterranean region in northern Spain. The study design includes three different contexts: sampling locations in which irrigation was initiated after the first two years of sampling ( $n = 9$ ), sampling locations in which irrigation was initiated after ten years of sampling ( $n = 8$ ), and sampling locations in which irrigation was never applied ( $n = 2$ ). We hypothesized that the implementation of irrigation would have strong effects across the avian community, with negative impacts on many or most of the farmland species. However, we also expected that some species may have responded positively to the changes brought on by irrigation. We thus expected structural changes within the bird community with potential consequences for conservation and management.

## 2. Materials and methods

### 2.1. Study area and data collection

Our study was conducted at 19 sampling locations across a 100 km<sup>2</sup> area (UTM 30TWN91; 42° 35' N, 1° 50' W; Fig. 1) located in the mid-western administrative region of Navarra (region V) in northern Spain. Region V of Navarra is an area comprised of 1300 km<sup>2</sup> and is characterized by a dry temperate Mediterranean climate, with average annual temperatures of 13.4 °C and rainfall of 560 mm. Land-use is dominated primarily by rainfed (i.e., non-irrigated) cereal farmland with native vegetation restricted to hillsides and ravines. Beyond cereal, there is variety of crops produced, including maize, fodder and vines (Table A.1, Appendix A). Official data shows that irrigated surface in the region grew by 124% between 2007 and 2019 (Table A.1, Appendix A). During that period, production of maize (exclusively an irrigated crop in the area) dramatically increased from 730 ha to 5508 ha. At the same time natural (grassland) and semi-natural (fallow land) vegetation drastically declined (a loss of almost 10,000 ha; Table A.1, Appendix A). Traditional management of fallow lands in the region involves leaving the stubble from harvest (end of summer) until the end of March; during this period, these fields are commonly used by livestock. From March onwards, fallow lands are usually ploughed, but the use of pesticides is not common, so vegetation grows regularly in fallow fields until sowing (following autumn). According to the study area's agrarian cooperative (Cooperativa Cerealista El Arga), the average expenditure on agrochemicals in irrigated crops (44.82 €/ha) is 70% higher than in rainfed crops (26.43 €/ha), which reflects the higher use of chemicals in irrigated crops than in rainfed crops.

Sampling was conducted by a single observer (D.V.) within the Common Birds Census Monitoring Program (SACRE, the acronym in Spanish) coordinated by SEO/Birdlife, using their standardized design and point-count survey protocols (Escandell, 2012). Sampling locations were chosen based on road and path availability, with a minimum distance of 1 km, and an aim to cover all habitats, and were thus not random. All 19 sampling locations (Fig. 1) were located in a rainfed agricultural environment during 2007, the first year of sampling. The 19 locations were sampled twice annually from 2007 to 2019 (except 2012 for logistical reasons), with the first sampling period occurring between 15 April - 15 May and the second between 15 May - 15 June. Surveys were always carried out during the early morning. During each point-count, all birds detected (heard and/or seen) within a five-minute period were recorded with no distance limits in the observations.

All 19 sampling locations experienced natural rain conditions between 2007 and 2008 (two years of sampling). In 2009, a new irrigation system was permanently installed in the eastern side of the survey area, such that nine of the sampling locations were irrigated from 2009 onward (Fig. 1). Another irrigation system was permanently installed in 2017 in the western side of the survey area, which led to an additional

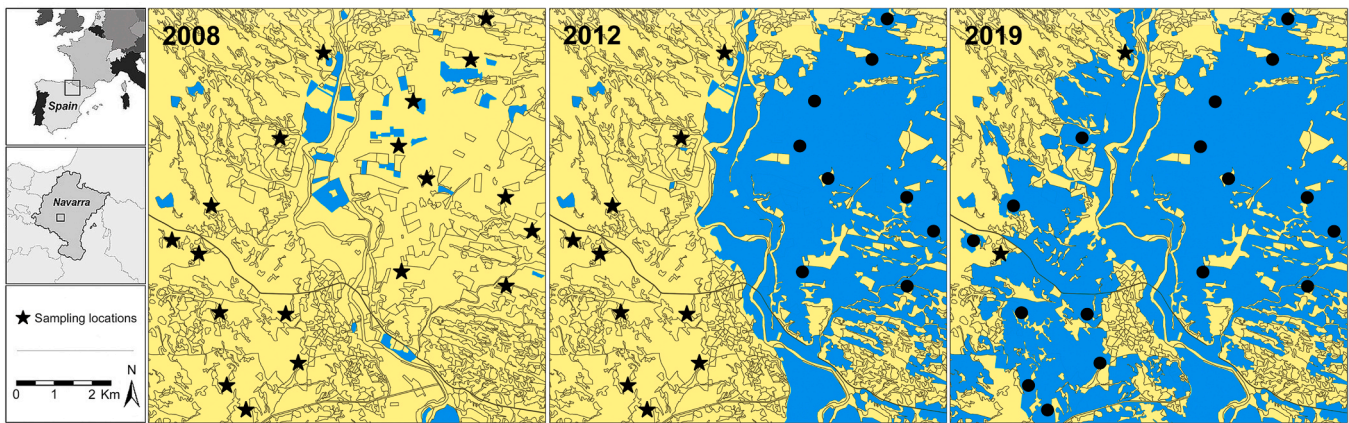


Fig. 1. Geographic location of the study area and sampling locations (black stars when non-irrigated and black dots when irrigated) at three different times over the course of the study: 2008 (left), 2012 (middle), and 2019 (right). Irrigated surfaces are shown in blue while rainfed surfaces are in yellow. At the beginning of the study (2008) all sampling locations were in rainfed farmland habitats. The different panels show the expansion of irrigation over time (<https://idena.navarra.es/Portal/Descargar>). (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

eight sampling locations being irrigated from 2017 onward (Fig. 1; Table A.2, Appendix A). Thus, only two sampling locations experienced no irrigation throughout the entire duration of the survey. The most common irrigation system in the area used non-mobile sprinklers, though drip irrigation systems were used in vineyards. Although we do not have precise information on crop changes in the 100 km<sup>2</sup> study area, personal observations by D.V. suggest that they are reflective of those in the region (see above, and Table A.1 of Appendix A), except for rapeseed, which is not abundant in the area. The cultivation of maize and rapeseed require different machinery and each agrarian cooperative has chosen to specialise in one over the other. Under irrigation, the traditional rainfed crops (barley, wheat, and vines) in this area transitioned to new irrigated crops (mostly maize, but also irrigated vines and to a much lesser extent irrigated legumes, sunflower and cereals). In the study area, maize is sown in late April, so at the time of the surveys the ground is bare or maize is only beginning to grow (not higher than 30 cm).

We detected a total of 120 species during sampling, but only included species with  $\geq 20$  detections, leaving 47 species for analysis (Table 1). Although the SACRE methodology may not be optimal to monitor certain species, we believe that the sampling protocol is adequate for our purposes because we use only presence-absence information (i.e., not species counts) and our modelling approach accounts for variation in detection probabilities among species (see Dorazio et al., 2006 for more information on the hierarchical community modelling framework).

For the presentation of results (i.e. post-hoc summaries across the study species) and to facilitate interpretation, we grouped species by their habitat preferences during the breeding period according to the large-scale monitoring program in which the data were collected (SACRE; SEO/BirdLife, 2019). Seven different habitat types are considered in SACRE: wetland, urban, shrubland, grassland, rocky habitats, agriculture (farmland), and forest. Species associated with a single specific habitat were defined as “habitat specialists”, while species associated with two or more habitats, or that were not attributed to a particular habitat, were classified as “non-specialists” (Table 1). Species habitat classifications can differ elsewhere in Europe, but the SACRE classifications reflect the most common habitat associations for species within Spain (SEO/Birdlife 2019). Among the 15 species classified as farmland specialists, the little bustard and the greater short-toed lark are also considered steppe birds, a species group of high conservation concern (Tucker and Evans, 1997). Both species are listed as vulnerable in Spain (Madroño et al., 2004) with little bustard categorized as vulnerable in Europe (BirdLife International, 2015) and threatened with extinction in Navarra (Decreto Foral 254/2019).

## 2.2. Statistical model

We estimated species occurrence probabilities, including the effect of irrigation, using a multi-species occurrence model (Dorazio and Royle, 2005; Zipkin et al., 2009) modified to accommodate a BACI design (Popescu et al., 2012). The BACI design allowed us to examine the effects of irrigation on individual species occurrence probabilities and localized species richness (Russell et al., 2009, 2015; Popescu et al., 2012). We accounted for imperfect detection using the replicate sampling occasions within years, which allowed us to separate a non-detection from a true absence (MacKenzie et al., 2002). We thus assumed closure between the two sampling periods within a year (no change in species occurrence status at sampling locations during the two months in a breeding season) but that species occurrence status could change annually.

We denoted the occurrence state (i.e. presence-absence) as  $Z_{i,j,t} = 1$ , if species  $i$  occurred at site  $j$  in year  $t$ . We modelled  $Z_{i,j,t}$  as a Bernoulli random process such that  $Z_{i,j,t} \sim \text{Bern}(\psi_{i,j,t})$  where  $\psi_{i,j,t}$  is the probability that species  $i$  occurred at site  $j$  in the year  $t$ . We modelled  $\psi_{i,j,t}$  with covariates using a logit link function:

$$\text{logit}(\psi_{i,j,t}) = \alpha 0_i \cdot (1 - irr_{j,t}) + \alpha 1_i \cdot irr_{j,t} + \alpha 2_i \cdot arable_j + year_{i,t} + site_j$$

The species-specific intercept was estimated based on a site’s irrigation status:  $\alpha 0_i$  is the intercept for species occurrence in the years before a site was irrigated ( $irr_{j,t} = 0$ ), while  $\alpha 1_i$  is the intercept in the years after a site was irrigated ( $irr_{j,t} = 1$ ). This approach allowed us to estimate the effect of implementing irrigation (and any subsequent consequences of irrigation) on baseline species occurrence (i.e., as  $\alpha 1_i - \alpha 0_i$ ; Popescu et al., 2012; Kéry and Royle, 2020). We included a species-specific effect of habitat ( $\alpha 2_i$ ) in which  $arable_j$  is the standardized percent of arable surface (cereal fields) within a 100 m radius buffer at each sampling location (mean = 77%, range 1 – 100), calculated using QGIS software (QGIS Development Team, 2018) and the 2012 Navarra’s land-use map (<https://idena.navarra.es/Portal/Descargar>). We used a buffer of 100 m radius because above this distance the detectability of most bird species is drastically reduced (Diefenbach et al., 2003). We verified that there was no variation in the proportion of arable land between years using the 2019 Navarre land-use map. Although we did not explicitly account for spatial autocorrelation, we incorporated a year by species random effect ( $year_{i,t}$ ) and a site-level random effect ( $site_j$ ) to account for unexplained variation across time and space.

We summarized the point count data in an array,  $Y_{i,j,t,k}$ , in which  $Y_{i,j,t,k} = 1$  denoted a detection of species  $i$  (1, 2,, 47) at site  $j$  (1, 2,, 19) during year  $t$  (1, 2,, 12) on replicate visit  $k$  (1 or 2). If a species was not

**Table 1**

A list of species included in our analysis, including their habitat classification and the estimated probability that their occurrence probabilities were lower after the implementation of irrigation. **Bold** indicates species with > 95%.

| English name            | Scientific name                  | Habitat preference | Number of detections | $P[\alpha I_i - \alpha 0_i < 0]$ |
|-------------------------|----------------------------------|--------------------|----------------------|----------------------------------|
| Little bustard          | <i>Tetrax tetrax</i>             | Farmland           | 28                   | <b>1.000</b>                     |
| Red-legged partridge    | <i>Alectoris rufa</i>            |                    | 79                   | <b>0.975</b>                     |
| Common quail            | <i>Coturnix coturnix</i>         |                    | 53                   | 0.767                            |
| Greater short-toed lark | <i>Calandrella brachydactyla</i> |                    | 39                   | <b>1.000</b>                     |
| Crested lark            | <i>Galerida cristata</i>         |                    | 229                  | 0.910                            |
| Calandra lark           | <i>Melanocorypha calandra</i>    |                    | 50                   | 0.636                            |
| Corn bunting            | <i>Emberiza calandra</i>         |                    | 252                  | 0.905                            |
| Black-eared wheatear    | <i>Oenanthe hispanica</i>        |                    | 43                   | <b>0.989</b>                     |
| Zitting cisticola       | <i>Cisticola juncidis</i>        |                    | 72                   | 0.508                            |
| European goldfinch      | <i>Carduelis carduelis</i>       |                    | 134                  | 0.036                            |
| Eurasian hoopoe         | <i>Upupa epops</i>               |                    | 85                   | <b>1.000</b>                     |
| European bee-eater      | <i>Merops apiaster</i>           |                    | 62                   | <b>0.994</b>                     |
| European turtle-dove    | <i>Streptopelia turtur</i>       |                    | 52                   | 0.586                            |
| Little owl              | <i>Athene noctua</i>             |                    | 28                   | <b>0.984</b>                     |
| Common kestrel          | <i>Falco tinnunculus</i>         | 67                 | 0.940                |                                  |
| Thekla's lark           | <i>Galerida theklae</i>          | Shrubland          | 57                   | 0.782                            |
| Tawny pipit             | <i>Anthus campestris</i>         |                    | 107                  | <b>0.975</b>                     |
| Woodchat shrike         | <i>Lanius senator</i>            |                    | 20                   | <b>0.968</b>                     |
| Iberian grey shrike     | <i>Lanius meridionalis</i>       |                    | 23                   | 0.877                            |
| Dartford warbler        | <i>Sylvia undata</i>             |                    | 53                   | 0.251                            |
| Subalpine warbler       | <i>Sylvia cantillans</i>         |                    | 67                   | 0.839                            |
| Sardinian warbler       | <i>Sylvia melanocephala</i>      |                    | 50                   | 0.055                            |
| European stonechat      | <i>Saxicola rubicola</i>         |                    | 64                   | 0.559                            |
| Melodious warbler       | <i>Hippolais polyglotta</i>      |                    | 26                   | 0.808                            |
| Red-billed chough       | <i>Pyrrhocorax pyrrhocorax</i>   |                    | Rocky habitats       | 38                               |
| Common wood pigeon      | <i>Columba palumbus</i>          | Forest habitats    | 22                   | 0.746                            |
| Common chaffinch        | <i>Fringilla coelebs</i>         |                    | 30                   | <b>0.975</b>                     |
| Great tit               | <i>Parus major</i>               |                    | 23                   | 0.903                            |
| Mallard                 | <i>Anas platyrhynchos</i>        | Water habitats     | 57                   | 0.629                            |
| White wagtail           | <i>Motacilla alba</i>            |                    | 55                   | 0.193                            |
| Common house martin     | <i>Delichon urbicum</i>          | Urban habitats     | 23                   | 0.234                            |
| Common swift            | <i>Apus apus</i>                 |                    | 52                   | 0.532                            |
| Common linnet           | <i>Linaria cannabina</i>         | Non-specialists    | 153                  | 0.300                            |
| European serin          | <i>Serinus serinus</i>           |                    | 51                   | 0.920                            |
| European greenfinch     | <i>Chloris chloris</i>           |                    | 47                   | 0.424                            |
| Cirl bunting            | <i>Emberiza cirlus</i>           |                    | 51                   | 0.492                            |
| House sparrow           | <i>Passer domesticus</i>         |                    | 81                   | 0.936                            |
| Barn swallow            | <i>Hirundo rustica</i>           |                    | 46                   | 0.245                            |
| Common nightingale      | <i>Luscinia megarhynchos</i>     |                    | 53                   | <b>0.984</b>                     |
| Common blackbird        | <i>Turdus merula</i>             |                    | 69                   | 0.477                            |
| Spotless starling       | <i>Sturnus unicolor</i>          |                    | 69                   | 0.864                            |
| Common cuckoo           | <i>Cuculus canorus</i>           |                    | 26                   | 0.376                            |
| Magpie                  | <i>Pica pica</i>                 |                    | 36                   | <b>1.000</b>                     |
| Carrion crow            | <i>Corvus corone</i>             |                    | 46                   | <b>0.995</b>                     |
| Western marsh harrier   | <i>Circus aeruginosus</i>        |                    | 34                   | 0.571                            |
| Black kite              | <i>Milvus migrans</i>            |                    | 23                   | <b>0.965</b>                     |
| Common buzzard          | <i>Buteo buteo</i>               |                    | 45                   | 0.546                            |

detected, then  $Y_{i,j,t,k} = 0$ . We modelled the detection-nondetection data by assuming that species detection was dependent upon the latent occurrence state of each species at each sampling location in each year:  $Y_{i,j,t,k} \sim \text{Bern}(Z_{i,j,t} \cdot p_{i,j,t,k})$ . The parameter  $p_{i,j,t,k}$  is the probability of detecting species  $i$  at site  $j$  in year  $t$  during survey replicate  $k$ , conditional on species  $i$  being present. Bird species detection probabilities are strongly affected by the sampling hour and date (Schmidt et al., 2013). To account for this variation, we included both hour and date as covariates on  $p_{i,j,t,k}$  using a logit link function:

$$\text{logit}(p_{i,j,t,k}) = \beta_0_i + \beta_1_i \cdot \text{hour}_{i,j,t,k} + \beta_2_i \cdot \text{date}_{i,j,t,k} + \beta_3_i \cdot \text{date}_{i,j,t,k}^2$$

$\beta_0_i$  is the intercept for species  $i$ .  $\beta_1_i$  is the effect of sampling hour, and the parameters  $\beta_2_i$  and  $\beta_3_i$  are the linear and quadratic effects of sampling date on the detection of species  $i$ . Hour was included as a linear effect because sites were only sampled during the morning while date includes both linear and squared effects because species detection

probabilities are expected to vary and peak at different times within the breeding season (Slagsvold, 1977). Both hour and date were standardized to have a mean of zero and standard deviation of one.

The species' models were linked by assuming that the intercept and covariate effects in both the occurrence and detection models ( $\alpha_0 - \alpha_2$  and  $\beta_0 - \beta_3$ ) were random effects, drawn from a community-level distribution (Kéry and Royle, 2015). For example, we assumed that the species-specific effect of arable land on occurrence probability ( $\alpha_2_i$ ) was drawn from a community-level distribution whose mean and variance were also estimated:  $\alpha_2_i \sim \text{dnorm}(\mu_{\alpha_2}, \sigma_{\alpha_2}^2)$ . We did not consider phylogenetic structure within the model. We estimated group-level mean occurrence probability, change in occurrence probabilities, and species richness post-hoc. We used a Bayesian approach for inference, carried out using JAGS (Plummer, 2003) and R (jagsUI R package; Kellner, 2016; R Core Team, 2019). We included uninformative prior distributions for the community-level parameters (see model code at Appendix B) and evaluated convergence by visually inspecting the trace

plots of monitored parameters and with the Gelman and Rubin convergence diagnostic (R-hat statistic value < 1.1; Gelman and Rubin, 1992; Gelman and Shirley, 2011).

### 3. Results

Irrigation had a negative effect on mean species occurrence (i.e.  $\mu_{\alpha 1} - \mu_{\alpha 0}$ ), although the 95% Credible Interval (CI) was large and overlapped zero (mean = -0.96, -3.03 to 0.95 95% CI; negative proportion of distribution: 0.70), revealing high variability among bird species in the community (Table 1, Fig. 2). Of the 47 species included in our analysis, irrigation had a strong negative effect on occurrence of 10 species (95% CI did not overlap zero) and a moderate negative effect on 16 species (50% CI did not overlap zero but 95% CI did). Another 16 species showed no response to irrigation (50% CI contained zero) while five species showed a moderately positive effect of irrigation (50% CI did not overlap zero but 95% CI did). No species had a strong positive

response to irrigation (Fig. 2). Irrigation also had a negative impact on species richness (Fig. 3) in both the eastern sampling locations after irrigation was implemented in 2009 (from 25.1 species pre-irrigation to 19.2 post-irrigation, on average) and the western sampling locations after implementation in 2017 (from 29.2 species to 22.7 on average).

Post-hoc analyses reveal that many species groups were negatively impacted by irrigation. The mean effect of irrigation was especially negative for farmland habitat specialists (mean = -1.63, -3.86 to 0.29 95% CI; negative proportion of distribution: 0.82). Irrigation had a strong negative effect on the occurrence of 6 of the 15 farmland species (Fig. 2), which led to a reduction in farmland species richness (from 11.1 species to 7.8 on average). In particular, four species experienced > 50% reductions in mean occurrence probability after irrigation was implemented (Fig. A.1, Appendix A): little bustard (from mean occurrence probabilities of 0.80 to 0.21), greater short-toed lark (from 0.53 to 0.09), Eurasian hoopoe (from 0.87 to 0.42) and black-eared wheatear (from 0.70 to 0.32). Irrigation also had a generally negative effect on

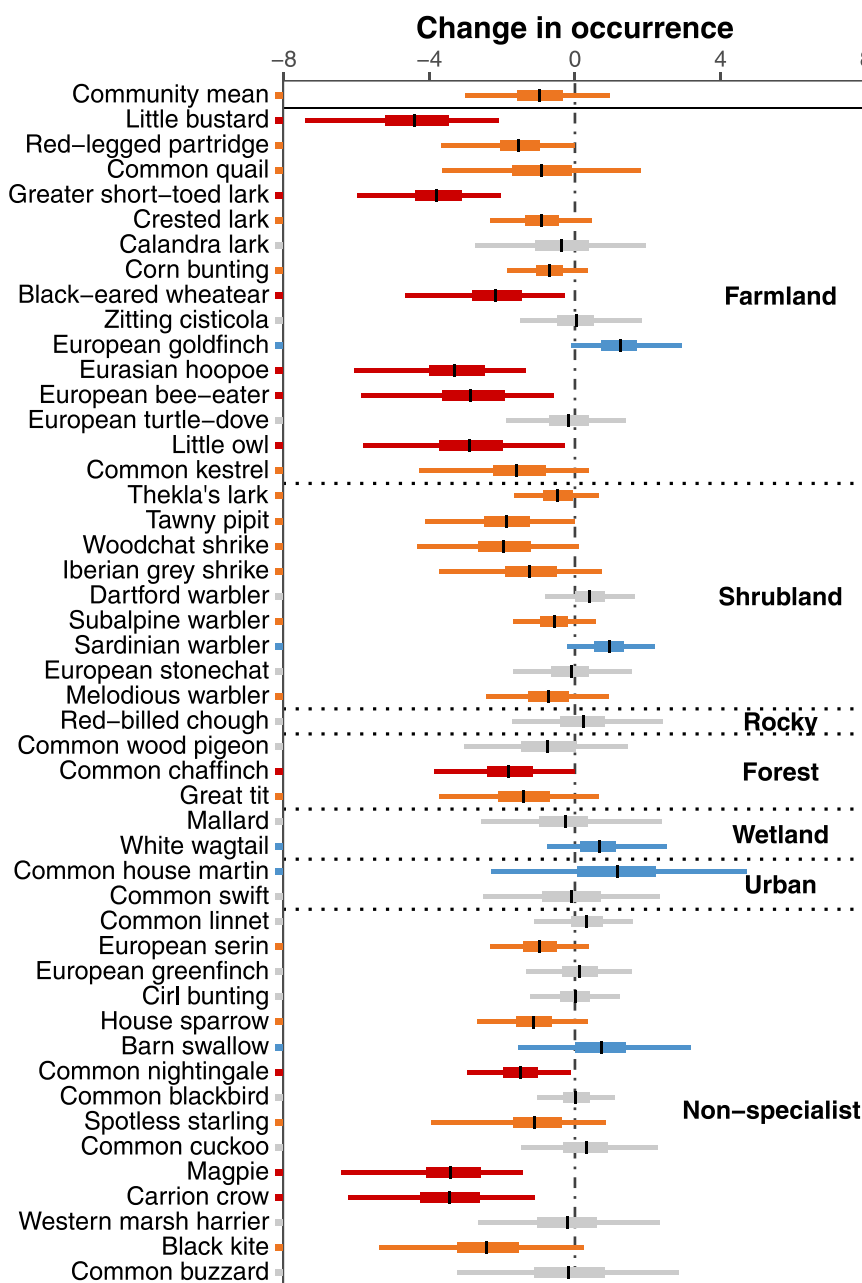
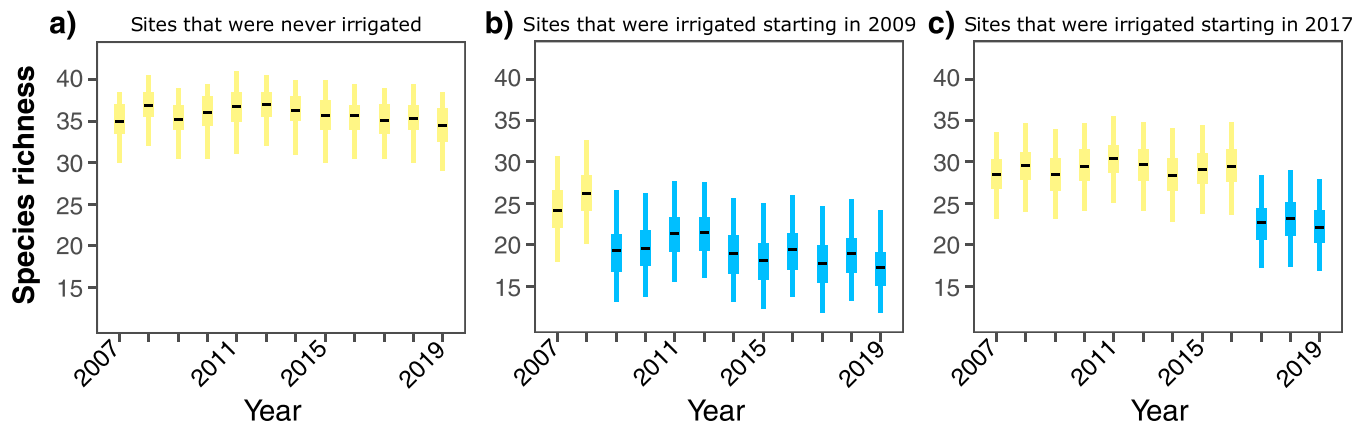


Fig. 2. Change in mean species' occurrences ( $\alpha_{1i} - \alpha_{0i}$ ) after irrigation for all 47 species included in our analysis. Species are organized by habitat classification: farmland, shrubland, rocky habitat, forest habitat, wetland habitats, urban habitat and non-specialist. The short vertical black lines show the mean change (across MCMC iterations), the boxes show the 50% credible intervals (CI) and the horizontal lines delineate the 95% CI. Light grey indicates no change in species occurrence probability, orange (negative effect) and blue (positive effect) indicate that the 50% CI does not overlap zero but an overlap of 95% CI with zero, and red (negative effect) shows that the 95% CI does not overlap zero. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)



**Fig. 3.** Mean site-level species richness at: a) sampling locations that were never irrigated (control;  $n = 2$ ); b) sampling locations in the eastern side of the survey area where irrigation was initiated in 2009 ( $n = 9$ ); c) sampling locations in the western side of the survey area where irrigation was initiated in 2017 ( $n = 8$ ). Yellow shows species richness in years in which sampling locations were not irrigated, while blue shows richness in years during which sampling locations were irrigated. The short horizontal black lines indicate the means, the boxes show 50% credible intervals (CI) and the vertical lines the 95% CI. No data were collected in 2012 and thus that year is not plotted in any of the panels. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

shrubland specialists, forest specialists, and non-specialist species (negative proportion of mean distribution: 0.68, 0.88 and 0.67 respectively; Table 1 and Fig. 2). The occurrence probabilities of six of the nine shrubland specialists, two of the three forest specialists and seven of the 15 non-specialists were at least moderately reduced (50% CI did not overlap zero; Table 1 and Fig. 2). As a consequence, these groups all suffered reductions in point-level richness: from 3.5 to 2.5 species (shrubland), from 1.4 to 1.1 species (forest), and from 8.3 to 6.3 species (non-specialist). Interestingly, five species experienced a somewhat positive effect of irrigation (Fig. 2) and these species did not belong to a single specialist group. One farmland bird (European goldfinch; from mean occurrence probabilities of 0.56 to 0.76), one shrubland bird (Sardinian warbler; from 0.10 to 0.22), one wetland bird (white wagtail; from 0.36 to 0.54), one urban bird (common house martin; from 0.49 to 0.71) and one non-specialist species (barn swallow; from 0.60 to 0.70), all showed a clear increase in their occurrence probability after the onset of irrigation (Fig. A.1, Appendix A).

Species-level occurrence probabilities were also strongly affected by arable land surface (community mean =  $-0.27$ ,  $-1.55$  to  $0.91$  95% CI). Overall, the amount of arable land cover had a positive effect on the occurrence probability of steppe birds, larks and six other species, while it had a negative effect on the occurrence probability of most other species (21 species; Fig. A.2, Appendix A). Many species are known to benefit from a mosaic landscape structure (Tucker and Evans, 1997) and thus a negative effect on species' occurrences was expected, as large arable land surface is associated with monoculture and habitat openness in this region. Detection probabilities (mean detection = 0.27) were also variable across species, with mean detection ranging from 0.09 for black kite to 0.69 for corn bunting. Across all species, detection probability was fairly consistent throughout the morning (Fig. A.3, Appendix A), as some species decreased in detection with time since sunrise (e.g. common quail, European turtle-dove, common nightingale), while others increased (e.g. little owl, common kestrel, Iberian grey shrike). Detection probability varied for most species by survey date, with average peak detection estimated to be around late May (Fig. A.3, Appendix A).

#### 4. Discussion

Our results demonstrate a clear effect of the implementation of irrigation and associated land use changes on bird community structure in a dry, Mediterranean landscape, with the majority of species (55%) showing a negative response in their occurrence probabilities and only a few (11%) showing a positive response. This led to an overall reduction

in site-level species richness by 24%. Moreover, among species groups, farmland birds, many of which are ground-nesters (Benton et al., 2003; Traba and Morales, 2019), were impacted most significantly (see also Giral et al., 2021), including negative effects on two steppe birds of high conservation value (Madroño et al., 2004; BirdLife International, 2015; Decreto Foral 254/2019). Other species groups, including shrubland species, forest species and non-specialists, were also negatively impacted by irrigation, suggesting that the effects of irrigation may be more widespread than previously thought. The striking changes in bird species occurrence probabilities reveal that irrigation may lead to fundamentally different bird communities at local scales, with communities in irrigated landscapes being more homogenous.

The implementation of irrigation – including associated effects and subsequent landscape changes – not only leads to increased water within local landscapes, but also results in the use of alternative crop species, increase of monocropping, land consolidation, loss of fallow lands, and a higher use of agrochemicals, all of which may be driving changes in local bird communities. In our study, farmland birds were most affected by the implementation of irrigation, likely because irrigation has a direct impact on their nesting habitat (i.e. the availability of cereal fields and fallow lands; Benton et al., 2003, Traba and Morales, 2019). For example, maize is frequently grown as an alternative to wheat and barley in irrigated arable land in Spain, but many farmland bird species avoid maize fields (Laiolo, 2005). When irrigation is implemented in vineyards, it typically leads to major changes of vine structures, altering avian habitats (Cabodevilla et al., 2021a, 2021b). In addition, irrigation also leads to more intensive farming and a more homogeneous landscape (Baldock et al., 1994, 2000; Giral et al., 2021). The increased use of agrochemicals can also negatively impact birds, either directly by affecting their fertility (Rodríguez-Estival et al., 2010; Lopez-Antia et al., 2016) or indirectly by reducing the availability of trophic resources (Geiger et al., 2010). The occurrence probability of some shrubland and forest birds as well as non-specialist species also decreased. Non-specialist species included species occurring in farmland as well as other habitats, so the process behind their declines might be similar to those of farmland species. In contrast, shrubland and forest species are unlikely to breed within or around the crops themselves, but irrigation may reduce the attractiveness of agricultural fields for foraging through a decrease in food abundance or availability as a result of changes in vegetation structure. However, the mechanisms driving negative changes within shrubland and forest species groups is less clear and should be investigated further. Alternatively, some species are likely finding increased resources in irrigated lands. Of the five species

showing positive responses to irrigation, three are species that tend to prey on flying insects (white wagtail, common house martin, barn swallow). These species may be finding higher densities of flying insects in irrigated landscapes due to the increased water available (Keiser et al., 2005; Jaleta et al., 2013). Despite potential positive changes for some species, the overall effect of irrigation was a loss of species richness. Thus, the direct and indirect effects of irrigation had a primarily negative impact on the local bird community, beyond farmland bird species that are typically considered in agriculture analyses.

Given the high potential for negative impacts on wildlife communities, irrigation schemes should explicitly evaluate approaches to mitigate biodiversity changes, especially for those species that are threatened. Ensuring the conservation of endangered farmland species requires a combination of management and policy actions to offset or limit the impact of widespread implementation of advanced irrigation strategies. Irrigation schemes should be implemented carefully and avoid the European Union's designated Special Protection Areas or areas with similar conservation value in non-European countries. Traditional rainfed agriculture with fallow lands should also be valued and actively encouraged as these systems offer unique habitat on which many bird species rely. When irrigation is implemented, maintaining some rainfed crops and fallow lands, varying the crops used within agrosystems, controlling the use of agrochemicals, and avoiding monocultures can lead to a heterogeneous landscape with resources for a range of species (Berg, 2002; Benton et al., 2003; Siriwardena et al., 2012). Fallow lands have been identified as critical for farmland bird conservation (Sanz-Pérez et al., 2019) because they provide foraging, mating, and nesting habitat for a number of species (McMahon et al., 2010; Schmidt et al., 2017; Traba and Morales, 2019; Tarjuelo et al., 2020). A mosaic structured landscape, in which fallow lands are dispersed within irrigated fields, is thus likely to reduce negative impacts on the bird community, although further research is needed including identifying the specific mechanisms causing changes.

There are several limitations to our study primarily associated with the fact that our experiment occurred naturally, rather than by design (i. e., we had no control over where or when irrigation was implemented). Because of this, there is a low number of control sampling sites ( $n = 2$ ) in which irrigation was never implemented. Further, we did not include spatial and phylogenetic structure within our modeling framework and thus, there may be confounding effects of these factors. However, despite these limitations, we estimated fairly strong effects of irrigation on the community of local birds, which suggests that the effects of implementing irrigation may be pervasive.

Our results provide evidence that the direct and indirect effects of irrigation on bird communities extends beyond farmland species, reaching many other bird groups. Although some species may respond positively to the changes associated with irrigation, we found that most species in our study system experienced reduced occurrence probabilities after the onset of irrigation. The implementation of irrigation is increasing worldwide in a race to increase farmland productivity (FAO, 2011). While irrigation can have a positive impact on crop production, it simultaneously has tremendous impacts on local ecosystems, including local bird communities, which may impact the biodiversity of other taxa and ecological processes more generally. Agricultural policies should ensure that food production is compatible with species conservation and the protection of local ecosystem services.

#### Authors' contributions

B.A., X.C. and D.V. designed the study. D.V. conducted the bird surveys. Statistical analyses were performed by X.C., with help from E. F.Z. and A.D.W. The initial draft of the manuscript was written by X.C. and E.F.Z., B.A. and A.D.W. contributed to subsequent drafts.

#### Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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