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Traditional small waterbodies as key landscape elements for farmland bird conservation in Mediterranean semiarid agroecosystems

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A B S T R A C T

Farmland bird populations are declining worldwide as a consequence of agricultural intensification, and the loss of singular landscape elements has been suggested as one of the main drivers. This scenario of agroecosystem simplification is even more exacerbated in arid and semiarid regions, where traditional small waterbodies (SWB) are rapidly vanishing due to groundwater overexploitation and the declining extensive pastoralism. Here, we used data from breeding bird surveys at SWB and adjacent control sites to assess for the first time the landscape-scale contribution of three types of traditional SWB for supporting farmland bird communities in the most arid region of continental Europe. Four community metrics were calculated for each SWB: species richness, abundance, diversity and proportion of conservation-concern species. In general, a high proportion (71% on average) of the local breeding bird communities used SWB, irrespective of the SWB type. Cattle ponds supported a greater abundance and proportion of threatened species, whereas drinking troughs were used by more diversified bird communities. Traditional artificial pools showed intermediate values for all community metrics. Our results support that in semiarid regions any type of traditional man-made SWB, if properly designed and managed, can play a pivotal role in supporting farmland bird communities at landscape scale. Despite their ecological importance, traditional SWB are often overlooked from agri-environment schemes, and their role for supporting farmland biodiversity is rarely considered. Therefore, effective SWB management and conservation measures should be implemented in the framework of the new reform of the European Common Agricultural Policy and other similar eco-schemes in order to halt the decline of farmland biodiversity.

1. Introduction

Agricultural intensification has caused significant declines in European farmland bird populations over the last century (Donald et al., 2001; Reif and Vermouzek, 2019). This change towards less nature-friendly farming practices has promoted the loss of extensive farm landscapes (Navarro and López-Bao, 2018), leading to semi-natural habitats (e.g., pastures and woodlots) and landscape elements

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(e.g., ponds, hedgerows and stonewalls) becoming less frequent in our modern agricultural landscapes (Concepción and Díaz, 2019). Such landscape elements, now called *Green and Blue Infrastructures* (GBI), are considered essential for farmland bird conservation, providing key resources for wildlife and acting as stepping stones for species dispersal within agroforest landscapes (Davies et al., 2016; Pustkowiak et al., 2021). As a consequence, the promotion of these GBI in farm landscapes must be a key pre-requisite for the successful development of agri-environment schemes (AES) (Concepción et al., 2012; Díaz et al., 2021), which are aimed to halt the decline of farmland biodiversity by compensating farmers for economic losses associated to the implementation of more nature-friendly farming practices (Reyne et al., 2020; Tarjuelo et al., 2021).

Following Biggs et al. (2005), we define small waterbodies (SWB) as standing waters between 1 m^2 and 2 ha with a maximum depth of no more than 8 m, which may be permanent or temporary and include both natural and man-made habitats. This definition includes a variety of waterbodies such as ponds, small lakes, farm ditches, artificial pools and drinking troughs (Biggs et al., 2016), among others. In recent decades, an increasing body of literature has stated the importance of SWB in providing both cultural, educational and recreational values as well as key ecosystem services, such as water storage for cattle or farming purposes, flood alleviation, fish farming or removal of nutrients or pollutants from water (Cheng and Basu, 2017; Fu et al., 2018; Hill et al., 2018; Oertli and Parris, 2019). In the same way, the important contribution of SWB in harbouring freshwater biodiversity has also been largely reported in recent literature (Bubíková and Hrivnák, 2018; Fait et al., 2020), especially in agricultural landscapes (Arntzen et al., 2017; Davies et al., 2008; Fuentes-Rodríguez et al., 2013).

In addition to their contribution for freshwater biodiversity conservation, SWB may be also highly relevant for adjacent ecosystem functioning through the provision of essential cross-system subsidies for wildlife, such as drinking water (Abdu et al., 2018a; Sutherland et al., 2018) or food resources (Lewis-Phillips et al., 2020). These services could benefit a broad range of farmland species, promoting more diverse communities, and improving some pivotal ecological processes such as pollination and seed dispersal in agrosystems close to SWB (Martínez-López et al., 2019; Stewart et al., 2017; Walton et al., 2020). Despite its interest, the ecological role of traditional SWB for farmland avifauna remains poorly studied in the literature, with the existing studies mostly corresponding to farm ponds in UK (Davies et al., 2016; Lewis-Phillips et al., 2019b) and game-related water troughs (Armenteros et al., 2021) or traditional SWB (Zamora-Marín et al., 2021b) in Spain. However, these studies used survey methods exclusively applied to the waterbody site, and comparative data on bird assemblages in the surrounding landscape have been rarely provided (but see Hanowski et al., 2006; Mckinney and Paton, 2009), thus precluding any inference on the relative importance of SWB to farmland bird conservation at landscape scale (Votto et al., 2020). Moreover, a comprehensive quantification on the importance of different structural types of traditional SWB to threatened birds is required to inform AES and wildlife management strategies (Armenteros et al., 2021; Lewis-Phillips et al., 2019a).

Despite their ecological and cultural values, SWB have been largely neglected from a conservation point of view worldwide (Hill et al., 2018), with a few exceptional cases from game management (see Lynn et al., 2008; Rosenstock et al., 1999). SWB are currently exposed to similar human-related threats than other types of water bodies, such as land drainage and changes in farming practices (Oertli et al., 2005). Moreover, most SWB from agricultural landscapes are of man-made origin, thus being additionally exposed to specific threats derived from an intensive or improper management (Zamora-Marín et al., 2021a). In these agroecosystems, traditional SWB have been historically created and actively managed by humans for extensive farming (i.e., farmland ponds or drinking troughs) and hunting purposes (i.e., water troughs targeting small game), thus being dependent from management practices. In recent decades, agricultural intensification has led to an unprecedented rate of SWB loss in several European countries (Ferreira and Beja, 2013; Hull, 1997; Reyne et al., 2020). This decline in SBW numbers may be even more exacerbated in water-scarce landscapes, such as agroecosystems from semiarid regions, where agriculture-mediated groundwater overexploitation is a major threat for freshwater conservation (Davis et al., 2013). Moreover, recent land use changes in these agroforest landscapes are promoting the abandonment and subsequent loss of traditional SWB (i.e., cattle ponds and drinking troughs), as a consequence of the steep decline experienced by livestock transhumance and other regimes of extensive pastoralism (Manenti et al., 2017). Thus, a research effort devoted to shed light on the importance of traditional SWB for farmland bird conservation is urgently needed for a successful AES implementation.

The aim of this study is to assess the landscape-scale contribution of traditional SWB for supporting farmland bird communities in semiarid agroforest ecosystems. In a previous study (Zamora Marín et al., 2021b), we used hierarchical models to estimate bird richness associated to three different types of SWB in southeastern Iberia, and highlighted differences in estimated avian richness related to SWB types. However, comparative data on the bird community inhabiting the SWB-surrounding ecosystem are needed for a comprehensive assessment on the role of SWB at landscape scale. Moreover, the importance of SWB to provide benefits for conservation-concern and declining bird species has been rarely quantified (but see Lewis-Phillips et al., 2019b), despite knowledge potentially generated by that research avenue could support the promotion of SWB as an effective management tool for threatened farmland bird conservation. Here, we used data from bird surveys at SWB and at adjacent control sites to calculate relative contribution of three SWB types to local bird communities, as well as to assess within-SWB differences in avian richness, abundance and conservation value. Based on previous studies (Lewis-Phillips et al., 2019b; Zamora Marín et al., 2021b), a high percentage of the bird community inhabiting SWB-adjacent ecosystems is expected to benefit from the presence of SWB through the landscape, as well as both SWB- and landscape-scale environmental variables are expected to affect bird richness, abundance and conservation value.

2. Methods

2.1. Study area

The study was conducted in the southeast of the Iberian Peninsula (province of Murcia; Fig. 1), which is considered the most arid region of continental Europe (Armas et al., 2011). This region covers an area of 11 317 km^2 and is characterized by a dry warm Mediterranean climate, with a strong water deficit during spring and summer. Despite the semiarid conditions, the study area shows a high ecosystem heterogeneity promoted by the existence of a coast-inland climatic gradient, which is expressed in the form of multiple environments varying in weather conditions, human pressure intensity, topography and the availability of water resources. Permanent watercourses (i.e., rivers or streams) are notably scarce and mostly restricted to the north-western corner of the study region, being absent from littoral areas and lowlands. Moreover, large artificial reservoirs are almost exclusively associated to these few permanent watercourses, and natural freshwater wetlands are extremely rare through the study region. Therefore, SWB are often the single water resource available for wildlife in most agroforest landscapes (Zamora Marín et al., 2021b). However, intensive irrigated agriculture has expanded to almost half of the study area during the last decades, thus promoting an excessive overexploitation of groundwater and surface water resources (Rupérez-Moreno et al., 2017). A more detailed description of the study area can be found in Zamora Marín et al. (2021b).

A total of 39 SWB spread over the province of Murcia (Fig. 1) was selected, belonging to three types in function of their structural features: traditional artificial pools, cattle ponds and drinking troughs. Traditional artificial pools ($n = 14$) were overspread on agroforest landscapes of the study area, where extensive agriculture and small game are the main land uses. They are permanent waterbodies with cemented bottoms and a round or square structure, and most of them are directly fed from small natural springs, while the rest are periodically provided with water by rangers or farmers. Because of traditional artificial pools are mostly devoted to extensive agriculture-related purposes and small game (mostly targeting the Red-legged Partridge *Alectoris rufa* and Turtle Dove *Streptopelia turtur*), they are continuously distributed through agroforest landscapes of the study region, from coast to inland zones. Cattle ponds ($n = 12$) are semi-permanent round waterbodies typical from Mediterranean farmlands. They were originally dug or traditionally modified to collect runoff water and provide drinking water for cattle. In the study region, they are exclusively located on steppe landscapes dominated by extensive grassland and almond or olive groves interspersed with small natural habitat patches. Despite their management or creation by humans, cattle ponds appear natural because of their silt bottom and absence of artificial structures around them. Lastly, drinking troughs ($n = 13$) are lineal permanent artificial SWB where cattle drink. They were originally created as a result from the modification of small natural springs by lining with cement to ensure water permanence, thus becoming artificial waterbodies. Contrary to cattle ponds, drinking troughs are exclusively located in mountain areas dominated by mosaic landscapes of Mediterranean mature forest and extensive farming.

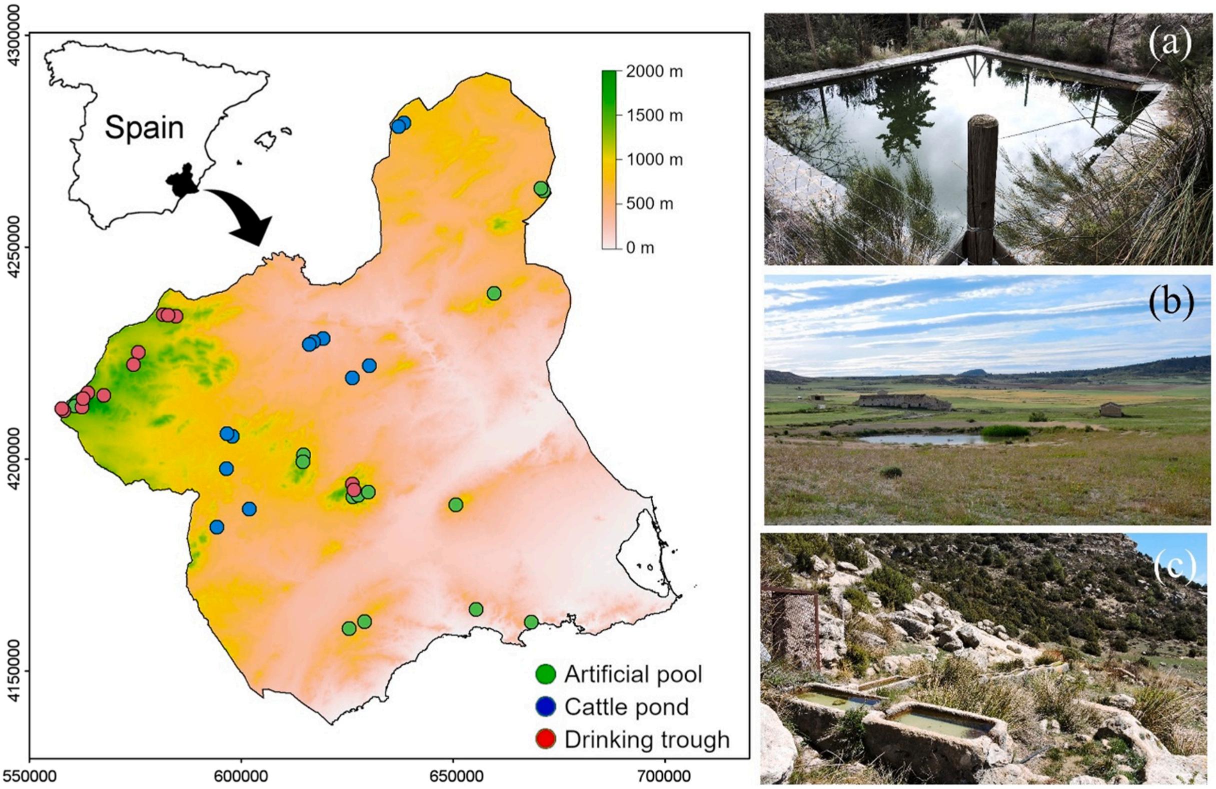

Fig. 1. Distribution of the 39 small waterbodies surveyed in the province of Murcia (southeastern Spain). Dot colour refers to the three selected waterbody types. UTM 30 S coordinates are provided in meters and the colour scale indicates the topography of the study area. A representative picture of each waterbody type is provided on right side: a) traditional artificial pool; b) cattle pond; and c) drinking trough.
Data and outline maps were obtained from public national data sources (https://www.ign.es/web/cbg-area-cartografia).

2.2. SWB and landscape features

Structural and environmental variables related both to the SWB and the surrounding landscape were recorded at each location (SWB plus adjacent sites). Selected variables were classified at SWB or landscape scale according to the spatial extent of their influence. Thus, six variables were recorded at the landscape scale (altitude, average annual rainfall, mean annual air temperature, habitat heterogeneity, terrain roughness and terrestrial vegetation cover), whereas other five variables were measured at the SWB scale (water surface, water depth, distance to the nearest waterbody, aquatic vegetation cover and surrounding emergent vegetation). Average rainfall and mean air temperature were extracted from the climate atlas of the province of Murcia (Garrido et al., 2013) with a 1-km^2 grid size. The free software Q-GIS (v 2.18.25) was used to calculate altitude, habitat heterogeneity, terrain roughness (TRI Index), terrestrial vegetation cover (NDVI Index) and distance to the nearest waterbody. Habitat heterogeneity was calculated as land cover heterogeneity following Hartel and von Wehrden (2013), by computing the standard deviation of the cover percentage occupied by four types of main land uses (woodland, Mediterranean shrubland, tree crops and herbaceous farming) in 1-km radius around study waterbodies. Data of land use types were obtained from the Spanish National Forestry Inventory (Dirección General de Desarrollo Rural y Política, 2012). A 1-km radius was also set to calculate the Terrain Roughness Index (TRI) and the Normalized Difference Vegetation Index (NDVI) with the aim of quantifying, respectively, topographic heterogeneity and terrestrial vegetation cover of the landscape where waterbodies were placed. A Digital Elevation Model was used to calculate the TRI index through Q-GIS (source Google Satellite: Imagery © 2018 CNES/Airbug, DigitalGlobe, Landsat/Copernicus). The selected buffer size is suitable to provide information on the general environmental conditions influencing the waterbody use by terrestrial wildlife (Eakin et al., 2018). Distance to the nearest waterbody was calculated as a proxy of the waterbody isolation degree (Boix et al., 2016; Cerini et al., 2020), by measuring the minimum distance from each study SWB to the nearest known waterbody with available surface water (watercourse, wetland or pond). The remaining variables were recorded in situ during bird surveys at each study SWB. The percentage of aquatic vegetation cover and surrounding emergent vegetation (immediately adjacent to SWB shoreline) were visually estimated always by the same researcher.

2.3. Bird surveys

From April to July, three visits were conducted to each SWB site with the aim of gathering an exhaustive list of associated bird species. Bird surveys in SWB were distributed into two annual cycles due to time constraints. Thus, 19 SWB were surveyed in 2017 and the remaining 20 were visited in 2018. Three surveys at each SWB were spread over the whole bird breeding season in the study area (April to July) to take a representative picture of the entire breeding bird community (Hanowski et al., 2006). The first visit was carried out in early-mid spring (April) to document early breeding bird species, such as resident warblers and some tits. The second visit was conducted on late spring (May-June), when peak-breeding season of migrant bird species occur, whereas the last visit was completed in mid-summer (late July) to document the later breeding species.

For each visit, a 3-hour census was conducted by direct observation from within a portable hide. Surveys began at sunrise (7:00–8:30) and were conducted in windless and rainless days. Portable hide was always deployed in the same position for each SWB and -at least- 10 m from water surface, thus maximizing the visibility of waterbody shoreline. The hide was set up close to surrounding vegetation to avoid effects on bird behaviour. In comparison to other classical bird techniques, this survey method provides higher detection effectiveness for bird species associated to SWB, as recently reported (Zamora-Marín et al., 2021a). Hence, an exhaustive inventory of bird species using each SWB was obtained by pooling the three visits to each site. All birds seen or heard within a 10-m buffer from the study SWB were recorded. Birds were identified at species level. Our 3-hour census period may imply double counting of the same individuals making use of a single SWB at different times, thus leading to abundance overestimation. However, surveyors were consistent in the bird counting procedure for all the study waterbodies, then any tendency to overestimate bird abundance would be equal across all sites.

In addition, bird surveys were also conducted in control sites adjacent to the studied SWB with the aim of recording data on local bird communities inhabiting the surrounding agroforest landscape. For that purpose, 1-km line transects were carried out along gravel roads and walking trails located at least 500 m away from study SWB. From April to July 2019, three transects were conducted at constant speed (2 km/h) in the surrounding habitat of each SWB site and under similar weather conditions than bird surveys at SWB. Transect direction and location was unchanged over the three survey visits (transects), as well as conducted by the same surveyors than bird surveys at SWB (JMZ-M and AZ-L). Transects began at sunrise (similar to SWB surveys), lasted 25–30 min and were 100 m fixed-width (50 m on each side). A laser rangefinder was used to avoid counting birds outside of the established bandwidth, thus all birds seen or heard within this buffer were recorded. It should be noted adjacent sites presented similar biotic and abiotic conditions (i.e., vegetation composition, vegetation cover, substrate type and altitude, among others) to SWB sites, being water presence the only difference between them. No riparian plant species were exclusively found in association to SWB sites, probably due to the small waterbody size (up to 432 m^2). Due to time constraints, bird surveys at adjacent sites were conducted in the successive breeding season than those from SWB sites. However, under unchanging habitat conditions, bird communities from Mediterranean agroforest areas are notably stable over time in terms of species composition (Herrando et al., 2003; Ukmar et al., 2007), and no habitat disturbances (i.e. fire or land-use changes) were observed in the surveyed sites over the study period (2017–2019). Therefore, composition of breeding bird communities was assumed to be stable.

2.4. Community metrics

Four community metrics on bird species associated to SWB were calculated: local richness, mean abundance, Shannon-Wiener diversity and proportion of conservation-concern species. Local bird richness (= alpha diversity for each SWB site) was calculated as the total number of bird species recorded at each site over the three visits. Bird abundance was averaged among the three visits to each SWB with the aim of making this parameter reliable. Shannon-Wiener diversity index was calculated through the *vegan* package (Oksanen et al., 2017). Though not included in our analysis, SWB-type bird richness (= gamma diversity) was computed as the total number of bird species associated to all SWB belonging to the same type. In addition to their positive impacts on farmland bird richness, AES design should take into account the guild structure of target groups by focussing also on threatened groups (Concepción and Díaz, 2011). Then, the conservation value of the recorded bird species was calculated to allow comparison of bird assemblages not only based on their diversity but also on the species conservation concern (Paquet et al., 2006; Pons et al., 2003). As species conservation priorities depend strongly on the considered spatial scale (Boyd et al., 2008), two conservation value indices were used following Paquet et al. (2006). Both indices account for the conservation status and abundance of the recorded bird species, but differ in their geographic coverage. The SPEC (Species of European Concern) index was calculated as a proxy on the conservation status of the recorded species at a broad-scale. This index is based on the European list of bird species conservation concern (BirdLife International, 2017), whereas an adapted regional index was calculated as a fine-scale proxy on the local conservation status using the regional red list of bird species (Robledano et al., 2006). Thus, an index value based in geometric progression of increasing conservation concern was assigned to each species (Table 1). The score was divided by the local richness with the aim of correcting for richness-related effects on the conservation value. For each SWB site, the conservation value was calculated for both indices as follow:

$$\text{Conservation value} = \frac{\sum_{i=1}^{s}[\log(A_i+1)\mathrm{x}\boldsymbol{Indexvalue}_i]}{s}$$

where s is the total number of bird species recorded at a given SWB site, A_i is the mean abundance of the species i in the considered SWB site and *Index value*$_i$ is the index value of the species i (see Table 1).

The contribution of each SWB site to support the local bird community was calculated as the proportion between the number of bird species associated to the considered SWB site and the number of species recorded in the location as a whole (i.e., overall richness considering together SWB and its adjacent site). Line transects at adjacent sites provided incomplete information on the local breeding bird communities, as highlighted by the fact that some bird species were exclusively observed at SWB sites for almost half of the paired SWB-adjacent sites comparisons (percentage of exclusive species at SWB sites: 31.6 ± 15.6 SD), thus precluding the use of line transects alone as an effective measure of the local breeding bird richness. Therefore, bird species recorded at each SWB site were pooled with those species observed at its adjacent site with the aim of making a complete inventory of the local breeding bird community inhabiting the surrounding landscape.

2.5. Data analysis

Principal component analysis (PCA) was conducted to assess the two-dimensional distribution of SWB sites based on environmental variables, as well as to explore relationship between bird metrics and environmental parameters. Non-parametric Kruskal-Wallis (hereafter K-W) H tests were conducted to explore differences in environmental variables and bird metrics among the three types of SWB. Then, when K-W tests yielded significant differences, Wilcoxon post-hoc tests were applied to make pairwise comparisons among the three studied SWB types. The same analytical procedure was also applied to assess differences in bird abundance and richness from line transects at adjacent control sites. True richness was also estimated at each SWB type by using the abundance-based estimator Jacknife-1 (Gotelli and Colwell, 2011) with the aim of considering sampling completeness. Two different bird survey methods (direct observations and line transects) were respectively applied at SWB and adjacent sites, thus precluding any comparison in terms of bird abundance between both sites. However, both methods are widely considered to provide representative data in terms of bird richness (Bibby et al., 2000; Zamora-Marín et al., 2021a), so richness comparison between SWB and adjacent sites was enabled to test the potential role of SWB in being used by terrestrial bird communities at landscape scale. All analysis were performed in R software v.4.0.3. (R Core Team, 2016).

Table 1
Allocation of an index value to the SPEC and IUCN regional categories, which are provided by BirdLife International (2017) and Robledano (2006), respectively. A geometrically increasing index value is assigned from lower to higher species conservation status. Bird species with no or unknown vulnerability status (LC, DD and NE) at regional scale are considered of no conservation concern (index value=1).

Index value	SPEC category	SPEC category description	Regional IUCN category*
16	–	–	CR
8	1	European species of global conservation concern	EN
4	2	Species of conservation concern and concentrated in Europe	VU
2	3	Species of conservation concern but not concentrated in Europe	NT
1	Non-SPEC	Species of no conservation concern	LC; DD; NE

* CR= critically endangered, EN=endangered, VU=vulnerable, NT= near threatened, LC= least concern, DD= data deficient, NE= no evaluated.

3. Results

SWB types significantly differed in all the structural, environmental and landscape features, except distance to the nearest waterbody (Table 2). Drinking troughs were placed in wetter areas and at higher altitudes, as well as presented lower depth in the water column. Cattle ponds were placed in areas with higher habitat heterogeneity but lower terrain roughness, terrestrial vegetation cover and surrounding emergent vegetation, whereas they showed larger water surface and higher percentage of aquatic vegetation cover. As highlighted by the PCA (Fig. 2), SWB sites were clearly clustered into the established typologies and there was a high intercorrelation between SWB types and environmental parameters. The main PCA axes represented ca. 62% of the total data variability, and axis 1 was strongly related with an aridity gradient, whereby SWB sites showing higher values of altitude, annual rainfall, terrestrial vegetation cover and terrain roughness (i.e., drinking troughs) were placed at the end of this axis.

A total of 91 bird species were recorded by considering together our surveys at SWB and adjacent sites. Taking into account the 39 studied SWB alone, 14,542 records of birds belonging to 80 breeding bird species and 34 avian families were documented (Supporting Information: Table S1). Recorded bird species corresponded to all existing SPEC and regional IUCN categories. This set of bird species associated to SWB comprised the 88% of the total pool of bird species detected over the entire study. Only 11 species recorded at adjacent sites were undetected at SWB (Fig. 3), which included taxa with very different ecological requirements (Supporting Information: Table S2). Conversely, eight bird species were exclusively detected at SWB, most of them corresponding to rare or extremely scarce species in inland zones of the study area, such as the Northern Goshawk (*Accipiter gentilis*), the hawfinch (*Coccothraustes coccothraustes*) and the Spanish Sparrow (*Passer hispaniolensis*). Overall, surveys at SWB sites allowed to detect the vast majority of bird species inhabiting adjacent agroforest landscapes. Paired comparisons among SWB-adjacent sites showed that more exclusive bird species (undetected at adjacent sites) were observed at SWB sites for cattle ponds and drinking troughs as compared to traditional artificial pools (Fig. 3). A clear linear relationship ($r^2 = 0.84$, $P < 0.001$) was observed between SWB and adjacent sites in terms of frequency of occurrence, then those bird species more common in adjacent sites were also more likely to be observed making use of SWB sites (Supporting Information: Fig. S1).

Differences in both bird community and conservation metrics were found among selected types of SWB. Both local bird richness and Shannon-Wiener diversity were significantly higher at drinking troughs, whereas no significant differences were found for both metrics between the remaining two SWB types (Figs. 4a and c; Table 3). Collectively, drinking troughs supported the highest gamma diversity (61 species), followed by cattle ponds (55) and traditional artificial pools (41). Jacknife-1 estimated richness showed the same pattern, highlighting that sampling completeness was similar and almost full for the three SWB types (traditional artificial pools: 95.7 %; cattle ponds: 85.9%; drinking troughs: 87.1%). In general, cattle ponds were the most relevant SWB type from a conservation point of view (Figs. 4d-4e; Table 3). However, significant differences were only found for the European scale-based index. At both scales, farmland bird communities associated to traditional artificial pools and drinking troughs showed similar conservation values. No significant differences were found in the local contribution made by SWB types to breeding bird communities. On average, each drinking trough and cattle pond provided benefits to 73.0 % and 72.7 % of the bird species inhabiting the immediately surrounding landscape, whereas the 68.0% was reported for traditional artificial pools (Fig. 4f; Table 3).

4. Discussion

4.1. Overall importance of SWB to farmland bird communities

Our study highlights the role of traditional SWB as key landscape elements for supporting farmland bird communities in Mediterranean semiarid agroecosystems. By comparing data from bird surveys at SWB and adjacent control sites, we provide new insights into the landscape-scale contribution of SWB to farmland bird conservation. Different SWB types provided benefits to a high proportion (on average, 71 %) of the breeding bird communities inhabiting the surrounding landscape, including several conservation-concern

Table 2
Mean and SD values for environmental and structural variables characterizing the three types of small waterbodies studied in the province of Murcia (southeastern Spain). Variables are pooled according to their effects at waterbody or landscape scale. *P-value* shows results from Kruskal-Wallis tests in relation to the differences observed in the selected variables among the three small waterbody types. Results from Wilcoxon pairwise comparisons are indicated with superscript letters beside each value. Same letters refer to groupings based on lack of statistical difference among waterbody types.

		Traditional artificial pools ($n = 13$)	Cattle ponds ($n = 12$)	Drinking troughs ($n = 14$)	*p-value*
Landscape	Altitude (m.a.s.l.)	854 ± 309^a	618 ± 191^b	1394 ± 136^c	< 0.001
	Average rainfall (mm/year)	415.4 ± 58.9^a	355.2 ± 23.7^b	525.1 ± 19.1^c	< 0.001
	Mean air temperature (°C)	15.18 ± 1.62^a	15.07 ± 0.95^a	12.42 ± 1.23^b	< 0.001
	Habitat heterogeneity	10.29 ± 10.36^a	25.05 ± 8.07^b	13.12 ± 7.33^a	< 0.001
	Terrain Roughness Index (TRI)	105.30 ± 42.94^a	28.54 ± 16.25^b	93.39 ± 27.03^a	< 0.001
	Terrestrial vegetation cover (NDVI)	0.20 ± 0.04^a	0.16 ± 0.04^b	0.23 ± 0.03^a	< 0.001
Waterbody	Water surface (m^2)	19.62 ± 14.12^a	267 ± 96.25^b	13.65 ± 13.96^c	< 0.001
	Water depth (cm)	56.64 ± 46.80^a	79.75 ± 35.42^a	12.08 ± 6.47^b	< 0.001
	Distance to nearest waterbody (m)	1052.9 ± 911.1^a	1059 ± 762.6^a	695.6 ± 390.3^a	NS
	Aquatic vegetation (%)	33.93 ± 35.25^{ab}	50.42 ± 21.26^b	17.31 ± 20.77^{ac}	< 0.05
	Adjacent woody vegetation (%)	24.64 ± 29.44^a	1.66 ± 3.89^b	26.54 ± 28.96^a	< 0.01

NS: non-significant.

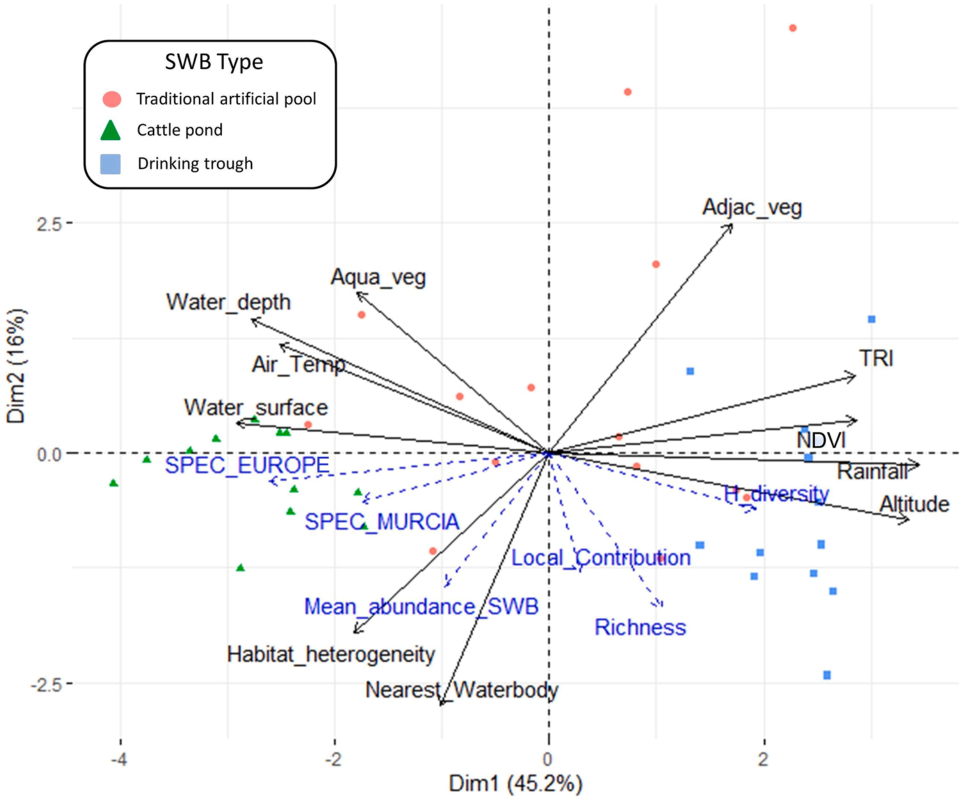

Fig. 2. Results from Principal Component Analysis (PCA) highlighting relationships among environmental variables, traditional small waterbodies (SWB) types and bird metrics in the province of Murcia (southeastern Spain).

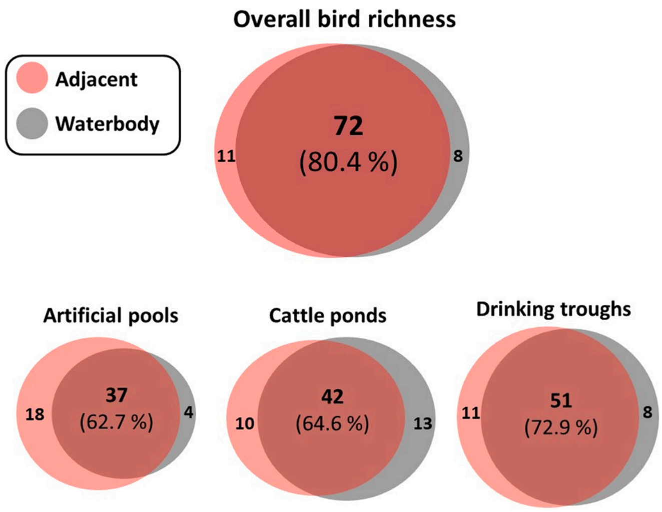

Fig. 3. Venn diagrams representing the number of exclusive and shared terrestrial bird species between adjacent and small waterbody sites for the total set of surveyed locations (upper diagram) and for the three selected types of small waterbodies (bottom diagrams) in the province of Murcia (southeastern Spain).

and threatened species (Supporting Information: Fig. S2). Such pivotal contribution is also expressed as high values of species richness and bird abundance, as well as the use by highly diversified bird communities, in line with recent studies (Abdu et al., 2018a; Lee et al., 2017; Lewis-Phillips et al., 2019a; Zamora Marín et al., 2021b). To our knowledge, richness values reported here are the highest ones available in literature concerning to terrestrial bird communities associated to SWB. In central Spain, a total of 54 bird species were

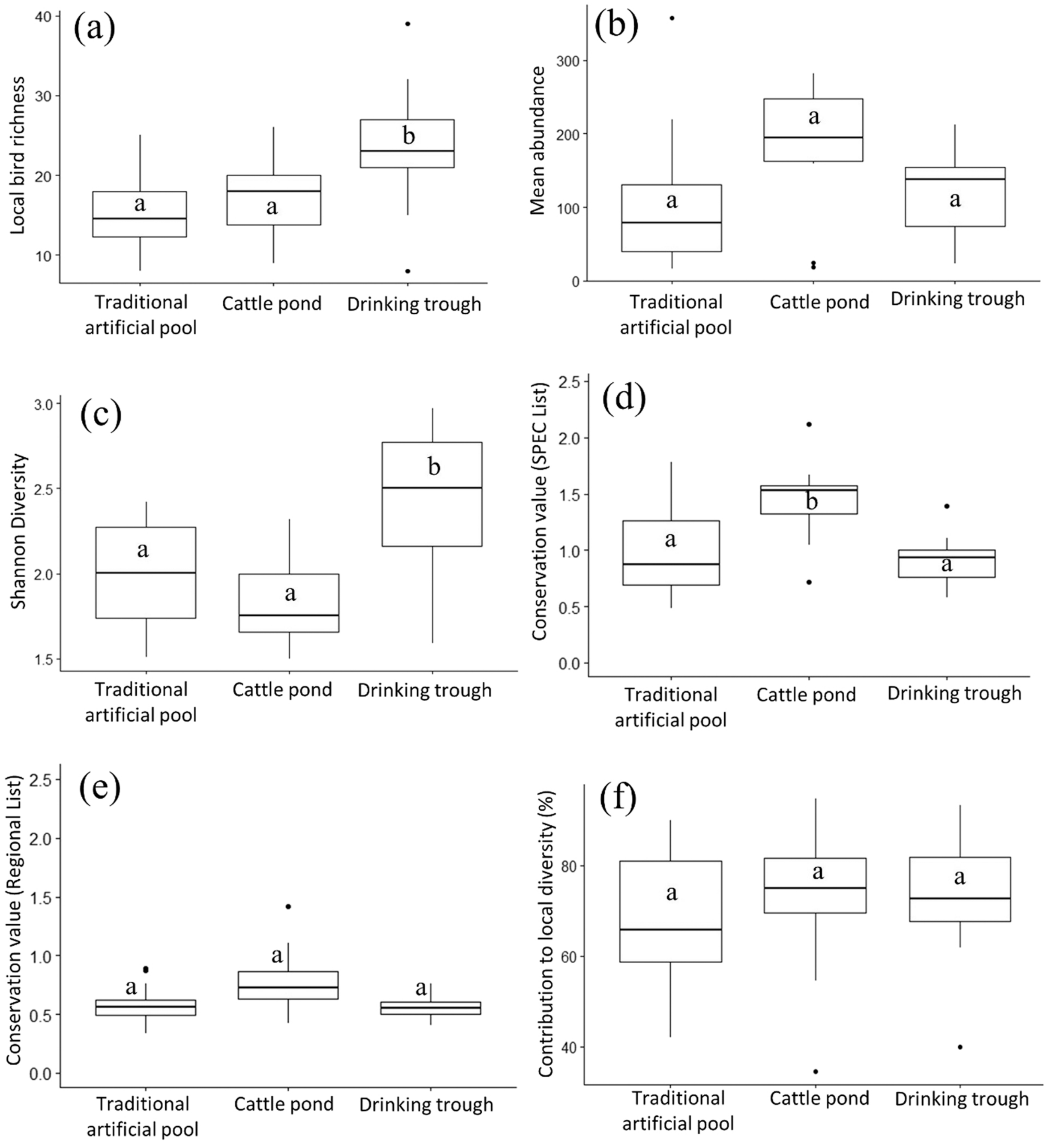

Fig. 4. Boxplots comparing bird metrics among three types of small waterbodies in the province of Murcia (southeastern Spain). Same letters within boxes (a or b) show groupings based on lack of statistical differences ($P < 0.005$) among small waterbody types, as stated by results from pairwise comparison Wilcoxon tests. Among-types differences for the selected bird metrics are showed as follows: (a) local bird richness (=alpha diversity); (b) mean abundance; (c) Shannon-Wiener diversity; (d) conservation value at European scale; (e) conservation value at regional scale; and (f) percentage of contribution to the local bird community.

associated to small-game water troughs as revealed by camera trapping (Armenteros et al., 2021). In the UK, 66 bird species were observed during the breeding season at 16 farmland ponds (Lewis-Phillips et al., 2019a), and slightly lower values were recorded for 25 ephemeral pools in forested landscapes from the USA (Scheffers et al., 2006; Silveira, 1998). In northeastern USA, 33 terrestrial bird species were recorded making use of 38 vernal pools (Eakin et al., 2018). Our study expands on previous research suggesting a great and overlooked importance of SWB to support farmland bird communities (Davies et al., 2016), particularly in Mediterranean agroecosystems (Armenteros et al., 2021; Zamora-Marín et al., 2021b). Unlike other really arid ecosystems, our traditional SWB in the

Table 3
Mean and SD values for bird metrics at the three waterbody types studied in the province of Murcia (southeastern Spain). Results from Wilcoxon rank sum test are indicated with superscript letters beside each value. Same letters refer to groupings based on lack of statistical difference among waterbody types.

	Traditional artificial pools ($n = 14$)	Cattle ponds ($n = 12$)	Drinking troughs ($n = 13$)	*p-value*
Local bird richness	15.36 ± 4.75^a	17.33 ± 4.81^a	23.69 ± 7.54^b	< 0.05
Mean abundance	105.0 ± 93.7^a	182.4 ± 85.0^a	120.9 ± 57.9^a	< 0.05
Shannon-Wiener Diversity	1.99 ± 0.30^a	1.85 ± 0.28^a	2.37 ± 0.46^b	< 0.05
Conservation value (Europe)	0.97 ± 0.37^a	1.44 ± 0.36^b	0.91 ± 0.21^a	< 0.05
Conservation value (Murcia)	0.59 ± 0.16^a	0.77 ± 0.28^a	0.57 ± 0.10^a	NS
Contribution to local richness (%)	67.99 ± 14.22^a	72.72 ± 16.15^a	73.01 ± 13.87^a	NS
Jack-1 estimated richness	46	64	70	–
Observed gamma richness	41	55	61	–
N species in Bird Directive	2	8	4	–
Sampling completeness (%) [estimated/observed richness]	95.7 %	85.9 %	87.1 %	–

NS: non-significant.

Iberian southeast have been historically present -though scarce- across the landscape (López Bermúdez et al., 2016), thus precluding the development of physiological responses by birds to become independent of drinking water (Zamora-Marín et al., 2021b). As a consequence, despite the limited size of our traditional SWB (< 500 m^2) and regardless of their typology, these free-standing isolated waters become essential landscape elements for farmland bird conservation in semiarid agroecosystems. Following the hypothesis that SWB promote bird diversity in temperate and semiarid regions, bird assemblages from strictly arid environments (i.e., deserts) –where water surface resources are completely absent- should be less diverse, but further research is required to uncover this question.

Interestingly, some bird species were exclusively detected at SWB sites and not in their corresponding adjacent sites (Fig. 3). This result may be explained by the fact that SWB deploy a pull effect on terrestrial wildlife, thus improving the detection probability of bird species through an increased waterbody use rate (Eakin et al., 2018; Zamora-Marín et al., 2021a). SWB may provide important ecological services for farmland bird communities at landscape scale and even driving the structure of bird communities at small spatial scales. Indeed, some bird species occur at higher densities when closer to SWB and even the distribution of certain species being spatially restricted to SWB (Abdu et al., 2018a; Borralho et al., 2008). Moreover, open ponds with high percentage of aquatic vegetation (i.e., cattle ponds) also provide high-quality food subsidies in the form of emerging insects (Declerck et al., 2011), then promoting the waterbody use by aerial insectivores (Lewis-Phillips et al., 2020). In our study, some of these species (i.e., the Barn Swallow (*Hirundo rustica)*, Common Swift (*Apus apus)* and Red-rumped Swallow (*Cecropis daurica)*) were observed doing foraging flights mostly at cattle ponds (Supporting Information: Table S1). Therefore, SWB could enhance the bird functional diversity in agroforest landscapes by promoting the occurrence of rare or habitat-specialist species, as reported for other singular landscape elements (García--Navas et al., 2022), but this question remain to be explored in further studies. On the other hand, bird surveys at SWB sites could be an effective complementary tool to get a more complete picture of the local breeding bird communities in semiarid regions, even allowing the detection of new terrestrial bird species unreported so far at regional scale (Baker et al., 2015; Zamora-Marín et al., 2021a).

4.2. Differences in bird communities among SWB types

Types of SWB differed notably in almost all of the measured environmental variables, as well as in some of the obtained community metrics. Further discriminant analysis intended to explore the influence of environmental variables on bird metrics were precluded by the fact that two SWB types were spatially clustered in the multidimensional space (Fig. 2), with cattle ponds appearing only in steppe lands (plateaus) and drinking troughs exclusively occurring in mountain areas from the northwestern corner of the study area (Fig. 1). This spatial configuration is explained by factors strongly related to traditional land uses, soil types, and the availability of natural springs across the study area. Therefore, both SWB types are unevenly distributed across the study area, so results from bird metrics must be interpreted as a joint response to the both SWB type and the environmental conditions of the surrounding landscape. In this regard, some ecological reasons may be inferred from the PCA results to explain the observed patterns among SWB types.

Bird richness and Shannon-Wiener diversity were significantly higher at drinking troughs. The use of this SWB type by a more diversified bird community could be explained by some contrasting environmental and structural attributes featuring drinking troughs. Drinking troughs are located at higher altitudes in wetter and more rugged areas, which are mainly embedded in a forest matrix with scattered patches of rainfed agriculture. This landscape configuration enhances among-habitat connectivity and promotes bird movement across different habitat patches. Moreover, drinking troughs show a great variety of micro-habitats, combining non-vegetated open shoreline zones with waterbody margins occupied by adjacent bushes. These features could allow the use of a diversified bird community (Davies et al., 2016; Lewis-Phillips et al., 2019a), with open-habitat birds (i.e. finches, buntings and doves) accessing waterbody through the non-vegetated shoreline whereas forest foliage gleaners (i.e. warblers and tits) accessing through the overgrown bush shoreline (Zamora-Marín et al., 2021b). Unlike drinking troughs, traditional artificial pools generally show structural conditions less prone to allow bird use (i.e., cemented bottom and vertical margins), thus hampering bird use. Conversely to drinking troughs, small ground-level puddles (filling from the main pool) rarely appeared in association to traditional artificial pools due to their greater water storage capacity, which precludes water overflow, and due to the maintenance works regularly conducted by stakeholders, thus avoiding water leakage from structural cracks. This feature could lead to traditional artificial pools being used by a less diversified bird community (Zamora-Marín et al., 2021b). Moreover, though placed in agroforest landscapes, most traditional artificial pools appear immediately surrounded by dense woody patches and even covered by closed tree canopy. Overgrown tree-shaded ponds have been recently reported to hold less diversified bird communities than open ponds (Davies et al., 2016; Lewis-Phillips et al., 2020). In turn, bird richness at adjacent sites of traditional artificial pools was slightly higher than in adjacent sites of cattle ponds (Supporting Information: Fig. S3), thus supporting our previous arguments about SWB-scale features make traditional artificial pools less prone to be used by terrestrial birds. In this regard, cattle ponds presented certain local conditions (i.e. ground-level access and non-vegetated shoreline) that seem suitable to allow use by strictly steppe bird species such as sandgrouse and some larks, which require clear views of open ground to drink safely and to minimize perceived predation risk (Abdu et al., 2018b; Ferns and Hinsley, 1995). However, these local conditions may also preclude the use of several bird species which prefer some extent of canopy or shrub cover to safely access SWB shoreline (Sánchez-García et al., 2012), thus making cattle ponds to be used by less diversified communities.

On the other hand, the greater bird abundance was recorded at cattle ponds, which might be due to the low availability of free-standing water resources in steppe areas (Hanowski et al., 2006; Silveira, 1998). In turn, bird abundance showed a clear positive relationship with distance to nearest waterbody, as highlighted from the PCA. Moreover, bird assemblages from steppe areas are mostly dominated by granivorous birds (Han et al., 2020; Traba et al., 2013), being this foraging guild widely reported to strongly rely on water intake for compensating the dry seed-based diet (Abdu et al., 2018a; Lee et al., 2017; Smit et al., 2019). Therefore, individual birds could have visited more frequently our cattle ponds, thus leading to abundance overestimation through double counting. In turn, adjacent sites to cattle ponds showed similar mean abundance values than those adjacent sites to traditional artificial pools and drinking troughs (Fig. S4), thus suggesting abundance differences were driven by an increased visitation rate in cattle ponds.

From a conservation point of view, SWB sites were visited by a non-negligible number of threatened or conservation-priority farmland bird species (Figs. 4d-4e; Table 3), most of them having experienced a strong decline over the last decades (Seoane and Carrascal, 2007; Traba and Morales, 2019). Five regionally threatened species were almost exclusively observed at cattle ponds, whereas bird species recorded at drinking troughs included only one threatened species and no threatened species were observed at traditional artificial pools. Cattle ponds supported also more conservation priority species than the remaining SWB types, as stated by the fact that these SWB were used by eight species listed in the EU's Bird Directive (Table 3). In this sense, steppe birds have been recently considered among the most threatened bird groups in Europe, mostly due to the fallowland loss in Mediterranean agroecosystems (Burfield, 2005; Traba and Morales, 2019). This situation is even more exacerbated in Spain, which harbours the largest European populations of many threatened farmland birds (Traba et al., 2007). Therefore, in face of the disappearing steppe landscape, Mediterranean cattle ponds must be considered as priority habitats for farmland bird conservation in Europe. Consequently, AESs should include specific measures to promote the restoration of the already existing cattle ponds, their effective protection and the new creation of these GBI in Mediterranean agroecosystems (Díaz et al., 2021). In this context, even too small waterbodies (i.e. water troughs targeting game) should be promoted in view of their potential to support a wide array of vertebrates (Armenteros et al., 2021), including conservation-concern steppe birds (Estrada et al., 2015).

Importantly, no significant differences were found into the contribution of the three SWB types to the local bird community, thus highlighting that SWB -whatever the type, but properly managed- are equally important to support farmland bird assemblages at landscape scale. SWB were visited by a high proportion of the breeding bird species in adjacent ecosystems (71% on average), and some sites showed values of local contribution even higher than 90%. However, bird richness data at SWB and adjacent sites were collected through two different survey methods, with line transects at adjacent sites involving less sampling time than direct observation at SWB. Hence, this aspect might lead to a slight underestimation of bird richness at adjacent sites, so this potential source of bias must be considered. Interestingly, despite the unsuitable conditions of some SWB sites (i.e., traditional artificial pools), they remained to be used by a great proportion of the local bird community. In this sense, our traditional artificial pools must be differentiated from the newly created modern irrigation pools, whose ecological contribution for overall biodiversity are notably low (Abellán et al., 2006; Sánchez-Zapata et al., 2005).

4.3. Synthesis and implications for conservation

This study provides novel contributions on the importance of traditional SWB for farmland bird conservation in Mediterranean semiarid agroecosystems. Traditional SWB studied here supported a high proportion of threatened and conservation-concern bird species, and made a great contribution to the local bird community through water provision, thus highlighting their role as key landscape elements in agroforest areas. These findings have important implications for farmland biodiversity conservation in water-scarce agroecosystems, such as those from other Mediterranean countries or semiarid regions. Moreover, our results may be used as starting point to currently wetter regions whose freshwater resources will be depleted under the ongoing global warming.

Despite their ecological importance, traditional SWB in the study area are rapidly vanishing due to the agricultural intensification, which leads to groundwater overexploitation and to the conversion to modern plastic-bottom irrigation pools (López Bermúdez et al., 2016; Rupérez-Moreno et al., 2017). Furthermore, this situation is expected to be strongly exacerbated in the study region by climate change effects through depletion of aquifer recharge (Pulido-Velázquez et al., 2018). Therefore, urgent conservation actions should be carried out in the framework of AESs to halt this decline and ensure an effective SWB protection. The new reform of the European Common Agricultural Policy (CAP) provides an excellent opportunity for promoting SWB restoration and creation in farm landscapes at continental scale, while accounting for the overlooked role of these freshwater habitats to terrestrial biodiversity. This new CAP reform sets minimum requirements to beneficiaries for devoting at least 3% of arable lands to non-productive landscape features, such as SWB, fallow lands and other unique farmland elements (Directorate-General for Agriculture and Rural Development, 2021). However, SWB-specific conservation measures should be considered in CAP and other AESs to maximize their potential for supporting biodiversity and to promote a diversified wildlife use. SWB-focused management practices will be particularly important for farmland biodiversity conservation in arid and semiarid regions, such those from the Mediterranean basin, where water scarcity makes SWB to take a critical importance for wildlife. According to the available evidence (Lewis-Phillips et al., 2020; Zamora-Marín et al., 2021b), these active management measures should include an effective pond restoration, creation of new near-natural SWB, good practices guidelines for pond users, establishment of minimum water levels to ensure standing water for wildlife, easy pond access by wildlife as well as the promotion of within-pond and landscape heterogeneity.

Declaration of Competing Interest

The authors declare that they have no conflict of interest.

Data Availability

The data that support the findings of this study are available from the corresponding author upon reasonable request.

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Author's contributions

J.M.Z.-M., J.F.C., F.J.O.-P. and D.S.F. conceived and supervised the entire study. J.M.Z.-M., A.Z.-L. and J.F.C. performed the fieldwork. J.M.Z.-M., J.F.C. and D.S.F. conducted the statistical analysis. J.M.Z.-M. wrote a first version of manuscript. All authors reviewed and contributed to the manuscript, as well as approved the final version.

Appendix A. Supporting information

Supplementary data associated with this article can be found in the online version at doi:10.1016/j.gecco.2022.e02183.

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Further reading

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