

## Terrestrial birds as indicators of agricultural-induced changes and associated loss in conservation value of Mediterranean wetlands

Francisco Robledano<sup>\*</sup>, Miguel A. Esteve, Pablo Farinós, M. Francisca Carreño, Julia Martínez-Fernández

Departamento de Ecología e Hidrología, Facultad de Biología, E-30100 Espinardo (Murcia), Spain

### ARTICLE INFO

#### Article history:

Received 10 February 2009  
Received in revised form 18 May 2009  
Accepted 25 May 2009

#### Keywords:

Wetlands  
Irrigated agriculture  
Steppe birds  
Indices  
Conservation

### ABSTRACT

During the last decades, agricultural intensification has modified the hydrology of Mediterranean wetlands, as has occurred in the Mar Menor coastal lagoon (SE Spain). Salt-steppe dominated wetlands, characteristic of transitional areas surrounding this lagoon and rich in biodiversity values, are threatened by changes in their water regime originated by land-use changes in the watershed. Traditional dryland cultures have also been replaced by irrigated ones. We assess the direct and indirect changes induced by agriculture on a terrestrial vertebrate community (steppe birds) especially sensitive to these ecosystem changes. This is made on the basis of several surveys of terrestrial birds (excluding aerial feeders and raptors) carried out between 1984 and 2008 in a representative wetland of the lagoon's continental margin (Marina del Carmolí). The changes in this bird assemblage reflect the hydrological modifications induced by agriculture at the watershed scale, which have significant effects on the relative representation of wetland habitats. Bird metrics and indices (species abundance, taxonomic composition, conservation value) describe these community changes as the combination of early declines in some species and families, and transient or late increases in other. In the long term, the family *Alaudidae* (and particularly species like *Melanocorypha calandra*) have lost importance to the benefit of *Turdidae* and *Fringillidae*. The area of salt-steppe explains a large part of the variation in the abundance of *Alaudidae*, while most variation of *Turdidae* and *Fringillidae* respond to the area of saltmarsh. Some *Alaudidae* seem to take advantage of the intermediate stages of saltmarsh expansion (*Calandrella rufescens*), or from the marginal irrigated crops fringing the wetland (*Calandrella brachydactyla*) that could compensate the loss of original agricultural habitats. Habitat changes in the wetland have occurred in three differentiated stages, and modify the steppe bird community towards a more heterogeneous assemblage including scrubland and palustrine species. Among three indices of ornithological value, only that based on the EU Bird's Directive Annex I species was negatively affected, but since the wetland has been designated a Specially Protected Area under this regulation, this represents a management failure. There exists some chance, however, to manage peripheral cropland in favour of biodiversity. The importance of monitoring in conservation evaluation and management is also stressed, since the terrestrial bird community of this wetland has not been regularly surveyed. In fact, its evaluation against the Bird's Directive criteria was made in a period of quick departure from the original, good ecological state of the wetland.

© 2009 Elsevier Ltd. All rights reserved.

### 1. Introduction

The importance of wetlands is increasingly recognised as systems supporting specific and valuable biodiversity (Maltby, 1986; Gibbs, 2000), and as areas playing a key role in essential ecological functions such as the control of sediment and nutrient flows and the removal of diffuse pollution at the landscape scale (Álvarez-Rogel et al., 2006; Moreno et al., 2007). This has promoted

different protection and conservation strategies which in the context of arid regions have an especial importance due to the singularity and key role of wetlands in their landscapes. However, conventional protection and conservation strategies usually do not take into account the close dependency of wetlands on the dynamics and management outside the protected area and this may interfere with the protection and conservation goals. Land use and management practices at the watershed scale affect the wetlands in many ways and especially through processes linked to the flows of water and nutrients reaching the wetlands.

In this context, we need to determine the extent to which wetlands react to land-use changes in the watershed, and how

<sup>\*</sup> Corresponding author. Tel.: +34 868 88 43 24; fax: +34 868 88 39 63.  
E-mail address: [frobleda@um.es](mailto:frobleda@um.es) (F. Robledano).

these changes can be tracked in different ecological compartments of the wetlands. We have tried to answer these questions studying a suite of biological indicators in a complex of wetlands in an arid landscape which has suffered important long-term land-use changes: The Mar Menor lagoon and its associated wetlands (Esteve et al., 2008; Pardo et al., 2008).

Urban changes, tourist development and especially the spread of irrigated agriculture favoured by the construction of the Tagus-Segura water transfer scheme in 1979, have led to a significant increase in the water and nutrient flows reaching the lagoon-wetlands complex. In the wetlands this has caused changes which are relevant at the complex scale, due to the role of the active wetland area in the removal of nutrients from diffuse sources, and also at the scale of the wetland itself, due to the effects on biodiversity. Understanding the dynamics of change is therefore important for conserving the biodiversity of the wetlands and for the sustainable management of the Mar Menor lagoon and watershed. The identification of indicator taxa is also a relevant issue for the monitoring of wetland changes, especially if these taxa share the properties of being ecosystem health and biodiversity indicators (Caro and O'Doherty, 1999; Duelli and Obrist, 2003). In this paper we focus in terrestrial birds as proximate indicators of habitat changes caused by irrigation and water drainage.

Birds have been used as bioindicators for many reasons, including the understanding of their ecology, the clearly demonstrated links among bird communities, vegetal associations, and territory, their coverage of different levels of the ecological pyramid in every environment, their detectability allowing rapid data collection not only on presence/absence but also on abundance (Padoa-Schioppa et al., 2006). Moreover, bird taxa are appropriate indicators for monitoring changes at ecosystem scale because (i) birds occur across a broad gradient of anthropogenic disturbance, from pristine wilderness to metropolitan areas; (ii) most birds live only a few years, so changes in species composition and abundance will manifest relatively quickly after a disturbance; and (iii) systematic and extensive bird surveys are currently conducted across many countries. Finally, birds are important to a large segment of the public, so the public may better relate to concerns about changes in bird communities than to those of other less popular taxa (Browder et al., 2002). In the European context, the advantages of birds as indicators, particularly with regard to agricultural intensification, have been pointed out by Gregory et al. (2005), and more recently Everard (2008) reaffirmed the value of bird based environmental indicators against other taxa using semiobjective criteria.

Surveys of bird populations and communities have been carried out in the Mar Menor lagoon and its associated wetlands since the early 70s of the past century. Previous pioneer studies (Guirao, 1859; Zamorano Ruiz, 1932) already recognise this as a rich area in terms of ornithological diversity. Monitoring effort, however, has been quite heterogeneous and still in recent years varies greatly in space and time. Long time-series of biological data are scarce, with the exception of wintering waterbird census more recently complemented by breeding waterbird population surveys. Typically terrestrial or palustrine bird communities (e.g. steppe and reedbed passerines) have been studied much less intensively, and good long-term records are scarce or totally lacking. In this paper we focus on terrestrial bird communities, and especially on steppe passerines which have the greatest conservation value in the European context (Tucker, 1997; Birdlife International, 2004; Santos and Suárez, 2005; Burfield, 2005). Although far from the systematic record of waterbirds, steppe passerines have the second best series of data among the potential ornithological indicators of landscape and environmental change in the wetland complex. We were able to reconstruct a series of data extending from 1984 to

2008, through the compilation of several surveys undergone in the same wetland area by the own authors and other researchers (Hernández, 1995; Torralva et al., 2003; Robledano et al., 2006), and to relate these with a comparable dataset on wetland habitat variation at the wetland scale. Since this variation has a well-documented relationship with agricultural intensification in the Mar Menor watershed, this allowed us to analyse the direct and indirect effects of agriculture (changes in the crop system and induced habitat changes) on this especially sensitive vertebrate community.

## 2. Study area and methods

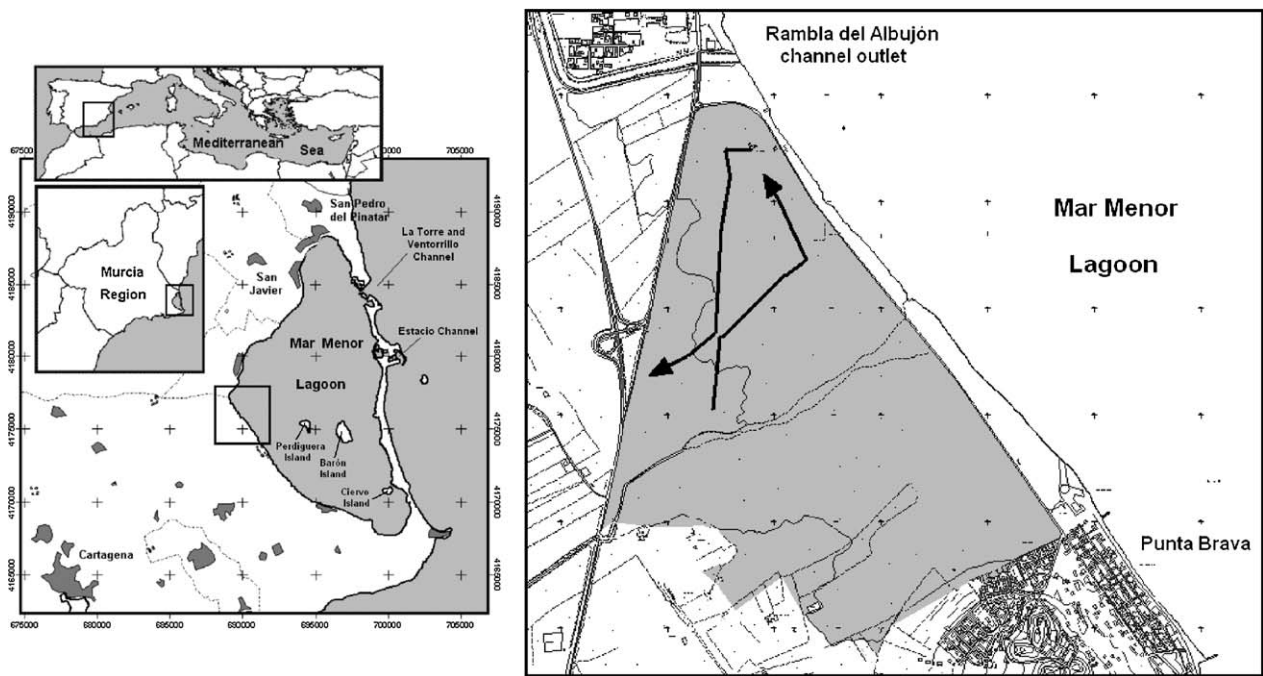
### 2.1. Study area

The Mar Menor lagoon is a hypersaline Mediterranean coastal lagoon located in Southeast Spain (Fig. 1). Ramsar Site since 1994, is the largest water surface of the western Mediterranean coast (135 km<sup>2</sup> surface area and a 580 hm<sup>3</sup> volume), and a remarkable biodiversity and scientific resource (EU Bird Specially Protected Area and Barcelona Convention's Specially Protected Areas of Mediterranean Importance since 2001). It is almost closed by a sand bar 22 km long with a very narrow connection with the open sea. Associated to its internal shore there are several margino-littoral wetlands, which are protected at both regional (Protected Landscapes) and international level (Ramsar and Barcelona Conventions, Natura-2000 Network) due to their natural and ecological interest. They include several natural habitats of priority and community interest according to the Habitats Directive (92/43 CEE). They are defined as coastal *crypto-wetlands* or "hidden seepages" (Vidal-Abarca et al., 2003). This paper refers to one of these wetlands, the Marina del Carmolí (Fig. 1), with a total surface area of 314 ha and located in the middle of the western side of the lagoon (latitude between 37°42'58" and 37°41'28"N; longitude between 0°50'25" and 0°51'58"W). Although most bird data were collected within the wetland itself, the objective was to analyse how the populations and assemblages of the site were influenced by agricultural change in the watershed, through its effects on the relative representation of wetland habitats.

The Mar Menor watershed is a 1200 km<sup>2</sup> plain slightly inclined towards this coastal lagoon and drained by several ephemeral watercourses (*ramblas*), which in natural regime flow towards the lagoon only after episodic storm rainfall events. Agriculture is the predominant land-use in the Mar Menor watershed. The discharge of the drainage system can be direct into the lagoon (surface outlets) or diffuse (subsurface flow through the peripheral wetlands). The spread of irrigation has caused both discharge types to increase. Direct ephemeral flows have become continuous water currents, while subsurface drainage has caused a general rise in the water table and the gradual conversion of steppe wetlands into palustrine ones. The area has a Mediterranean arid climate, with warm winters, an annual mean temperature about 17 °C, annual mean rainfall of 330 mm and a high inter-annual rainfall variation. The Carmolí wetland lies at the end of the largest sub-basin (633 km<sup>2</sup>) of the Mar Menor watershed, accounting for more than half of its total surface.

### 2.2. Habitat changes

From 1984 to 2001, temporal and spatial changes in total surface area and internal habitat composition of the main wetlands fringing the inner (continental) shore of the Mar Menor Lagoon were studied by means of remote sensing, in order to analyse their conservation implications, especially as regards the application of the Habitats Directive. All these wetlands present salt steppes, saltmarshes, reedbeds and sandy areas, although with a different



**Fig. 1.** Map of the Mar Menor Lagoon, showing the location and present extension of the study area (Marina del Carmolí), as well as the routes used to survey terrestrial passerines in 2008. The northern section of the wetland was selected in order to avoid palustrine flooded areas.

relative importance. Following the typology of the Habitats Directive, the sandy units are mostly composed of the following habitats: 1210 “Annual vegetation of drift lines” and 2210 “Fixed beach dunes with *Crucianellion maritima*”. The salt-steppe units are 95% composed of the priority habitat 1510 “Mediterranean salt steppes, *Limonieta*”, whereas the remaining 5% comprises habitat 1430 (Halo-nitrophilous scrubs *Pegano-Salsoletea*) and 92D0 (Southern riparian galleries and thickets). Main species in salt steppe are *Lygeum spartum*, *Suaeda vera*, *Frankenia corymbosa* and *Limonium caesium*. The saltmarsh unit is dominated by habitat 1420 (Mediterranean and thermo-Atlantic halophilous scrubs, *Sarcocornetea fruticosi*), although there are also small patches of habitat 1410 (Mediterranean salt meadows). Main species in saltmarsh are *Sarcocornia fruticosa*, *Arthrocnemum macrostachyum*, *Halimione portulacoides* and *Limonium cossonianum*. Finally the reedbeds unit is dominated by *Phragmites australis*. All habitats are recognised as being of Community Interest with the exception of habitat 1510 (Mediterranean salt steppes), which is designated as of Priority Interest. Reedbeds are not included in the Habitats Directive.

Land cover maps for the wetlands were obtained by photo interpretation of aerial photographs and supervised classification using the widely used maximum likelihood algorithm (Michelson et al., 2000; Richards, 1995). Landsat images sensors TM and ETM+ covering 5 years of the period 1984–2001 (1984, 1992, 1995, 1997 and 2001) were used. Each classification was carried out with two images (summer and winter) and the NDVI (Normalised Difference Vegetation Index) for each image. The images were processed with GRASS (Geographic Resources Analysis Support System, <http://grass.itc.it>) an open GIS system under Linux. The methodology was verified by cross-validation using a stratified random sampling. Overall accuracy percentage of sampled pixels which are well classified reaches 85% in Marina del Carmolí.

The following land cover classes were identified (Carreño et al., 2008): natural vegetation (three units: saline steppe, saltmarsh and reedbeds); agricultural fields (irrigated cultures); water bodies (mainly portions of the Mar Menor lagoon); bare ground (inactive

ponds in saltworks, river beds in ephemeral channels); and infrastructures (urban settlements, roads, rubbish tips). The narrow strips of sandy areas are included in the nearest vegetal units. These maps have allowed to track the changes in the area and in the internal composition of wetlands between 1984 and 2001 (Table 1 and Fig. 2).

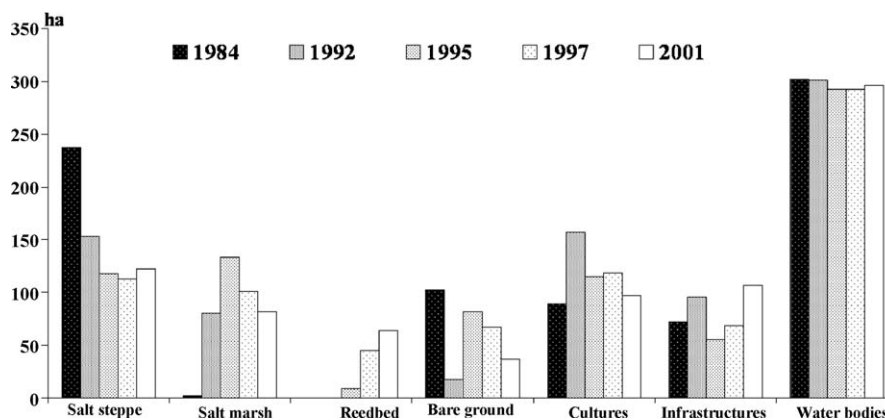
### 2.3. Bird data

Census made along line-transects were used to track changes in the terrestrial bird communities of the Marina del Carmolí wetland (Fig. 1). This wetland has been the subject of research on aspects like soils and vegetation, water, nutrient and pollutant dynamics (e.g. Álvarez-Rogel et al., 2006, 2007; Jiménez-Cárceles and Álvarez-Rogel, 2008). But, despite steppe birds are the key biological value supporting its designation as a Specially Protected Area (SPA) under the Birds' Directive, there is no published research on this topic. Particularly, we lack information on how habitat changes caused by agricultural intensification in the watershed are affecting the terrestrial passerine assemblage. Fortunately, the area has been included in several academic and conservation-oriented programmes of research and monitoring. This allowed us to gather abundance indices (IKA = Index of Kilometric Abundance, or birds/km) for all species of the terrestrial community. These were recorded in 7 years of transect sampling (spread along a period of 24 years): 1984, 1989, 1995–1997, 2003, 2008. Additional data on the bird assemblage living in dryland

**Table 1**

Changes in the relative surface area of the main types of habitats within the 400 ha square containing the Marina del Carmolí wetland, quantified through remote sensing between 1984 and 2001. Only the four main types are shown.

Habitat type	1984	1992	1995	1997	2001
Saline steppe	237.23	153.43	118.25	112.5	122.19
Saltmarsh	2.03	80.44	133.38	101.06	81.56
Reedbed	0	0	8.56	44.75	64.13
Cultures	89.38	156.75	115.31	119	97.38



**Fig. 2.** Changes in the surface area of the main types of habitats within the 400 ha square containing the Marina del Carmolí wetland, quantified through remote sensing between 1984 and 2001 (after Carreño et al., 2008).

(herbaceous) agricultural fields were available only for the year 1984. This steppe-like habitat located around the wetland was surveyed following the same procedure (3 winter and 5 summer bird census). After this year these cultures were substituted by irrigated crops, abandoned or colonised by wetland vegetation. The results of these surveys were included in multivariate analysis of the bird communities, since they are thought to represent a original type of terrestrial bird assemblage (like that of saline steppe) not affected by irrigated agriculture. Seasonal averages (for summer = April–September, and winter = October–March) have been calculated when more than one transect census was available. The overall composition of the community under study included species and/or families representing the various habitats present.

Line-transects had a length of 0.5–1 km depending on the source sampling scheme, with the outer limit of the counting strip usually set at 40 m, on both sides of the transect. When a smaller sampling strip was used (20 m in 1995–1997), the results were corrected to refer to similar surface areas. Data for 2003 (Torralva et al., 2003), given in birds per hectare, were also made comparable by referring them to an equal surface of that covered by the transects. On the other hand, by expressing the results of all surveys in birds/10 ha, we were also able to compare local densities with those recorded elsewhere, and to use their temporal variation as an index of change in ornithological value. Sampling was carried out by one or two observers who followed a pre-established route at a more or less constant speed (approximately 2 km/h), always during the first 4 h of daylight. The Marina del Carmolí wetland keeps a network of ancient paths, closed to vehicles and scarcely used by pedestrians, but distinguishable enough to facilitate the bird surveys. We believe that the fragmentation or edge effects caused by these paths are minimal due to their partial colonisation by vegetation. Although we cannot determine the exact sampling places during all the study period, we are confident (and most researchers involved in sampling have confirmed so) that the routes followed are representative of the average conditions of the terrestrial phase of the Carmolí wetland. A small part of the variation, however, can be attributable to observer or methodological bias, and the results have to be interpreted cautiously.

Sampling effort ranged between a single seasonal survey per year (1984) to 10–11 surveys per season (1995), and on average approached bi-monthly surveys (5.3–5.6 surveys/season). The final series consists approximately of a survey every 5 years. We used several indices of abundance and community structure to relate changes in passerine populations and taxocenoses to habitat and landscape variables.

#### 2.4. Statistical analyses

The resulting raw matrix of abundance data consisted of 83 rows (sampling dates)  $\times$  35 columns (species). Aerial foragers were not included in the species list, as well as those species appearing only on 1 year. Separate analyses were performed on winter and summer abundance matrices (41  $\times$  27 and 42  $\times$  27, respectively). To show general trends, we averaged abundance values (IKAs) for 6 years (1984, 1989, 1995, 1997, 2003 and 2008). Also on a yearly basis, we summed the IKAs of the most important families in terms of abundance and habitat specificity. We also computed total community abundance, species richness and diversity (Shannon–Wiener index). In order to detect changes in the conservation value of avifauna, we also calculated indices of conservation status, adapted from the procedure used by Pons et al. (2003), ranking species according to their inclusion in Birds in Europe SPEC categories (Birdlife International, 2004), IUCN Spanish Red Data Book (Madroño et al., 2004) and Annex I of the EU Bird's Directive. These values were multiplied by the abundance index (IKA) logarithmically transformed (Pons et al., 2003; Paquet et al., 2006).

Yearly abundances, richness, diversity and conservation status indices were related through generalized linear regression models (GLM) with four predictor variables describing the relative representation of habitats, in turn related with the land-use changes in the watershed and the consequent *humidification* of the Carmolí steppe wetland (Carreño et al., 2008; Esteve et al., 2008; Table 1). These variables were the surface of saline steppe, saltmarsh scrub, reedbed and irrigated cultures in a 4 km<sup>2</sup> square window containing the wetland. These are thought to illustrate the general landscape setting (coverage of different habitats) but also habitat structure (vertical and horizontal complexity). Mean IKA of passerine species and families, and the community structural indices for 1984, 1989, 1995, 1997 and 2003 were regressed on habitat variables for 1984, 1992, 1995, 1997 and 2001 (data not available thereafter). Although data points are not exactly coincident, we consider them representative of consecutive stages in the wetland ecosystem with an approximately 5 year spacing. Data were analysed with the statistical package R (The R Foundation for Statistical Computing, 2005; <http://www.r-project.org>).

To get a general picture of temporal change within the terrestrial bird community, we performed multivariate analysis on the raw matrices. Individual samples (monthly census) were submitted to classification and ordination analyses based on the similarity of their passerine bird communities. We used Multi-dimensional Scaling (MDS) to graphically represent the samples and interpret the temporal gradients, using similarity percentages to delineate homogeneous periods (Mander et al., 2007). Once

these stages were defined, we conducted an indicator species analysis for both summer and winter communities, using the SIMPER procedure, which identifies those species contributing most to the Bray–Curtis dissimilarities between groups of samples (Jansen and Robertson, 2001). All these analyses were implemented through the PRIMER 6 statistical package.

### 3. Results

In summer, total community abundance (1984 samples from dryland cultures excluded) declined during the 24-year period of study (adjusted  $r^2 = 0.6$ ;  $n = 6$ ;  $p < 0.05$ ), but species richness and Shannon diversity index increased. Diversity values, in any event, were initially and on average lower than those reported in the literature for this same site and for similar ecosystems (Hernández, 1995). *Alaudidae* was the only family showing an overall significant decline with year (adj.  $r^2 = 0.81$ ;  $p < 0.01$ ), while there was an increase in the proportional abundance of *Fringillidae*, *Turdidae* and *Sylviidae* (Fig. 3).

In winter, a similar pattern appears for most families and structural indices, except that total community abundance appears

more stable (or slightly increasing), and the overall contribution of 'other' families, mainly *Motacillidae* and *Emberizidae*, is greater (as corresponds to a richer community). Shannon  $H'$  and richness increase significantly with year (adj.  $r^2 = 0.97$  and  $0.72$ ;  $p < 0.001$  and  $0.05$ ). The overall decline of *Alaudidae* is less pronounced than in summer yet also significant (adj.  $r^2 = 0.77$ ;  $p < 0.05$ ), and there is a remarkable transient increase of *Fringillidae* in 1995–1997 (Fig. 3).

Not all members of the steppe community respond individually to habitat variables, but representative species and families that can be considered habitat specialists do so (Table 2). Figs. 4 and 5 show some examples.

The MDS ordinations, performed individually for winter and summer matrices (Fig. 6), identify three more or less defined periods between which compositional and structural change occurs in the community. The first phase of the study (1984–1989) is characterized in the wetland by a dominance of the typical saline steppe community, well differentiated from the assemblage of remnants of dryland crops (all 1984 samples except one). During the second phase (second half of the 1990s) the influence of irrigation has caused the gradual replacement of the steppe

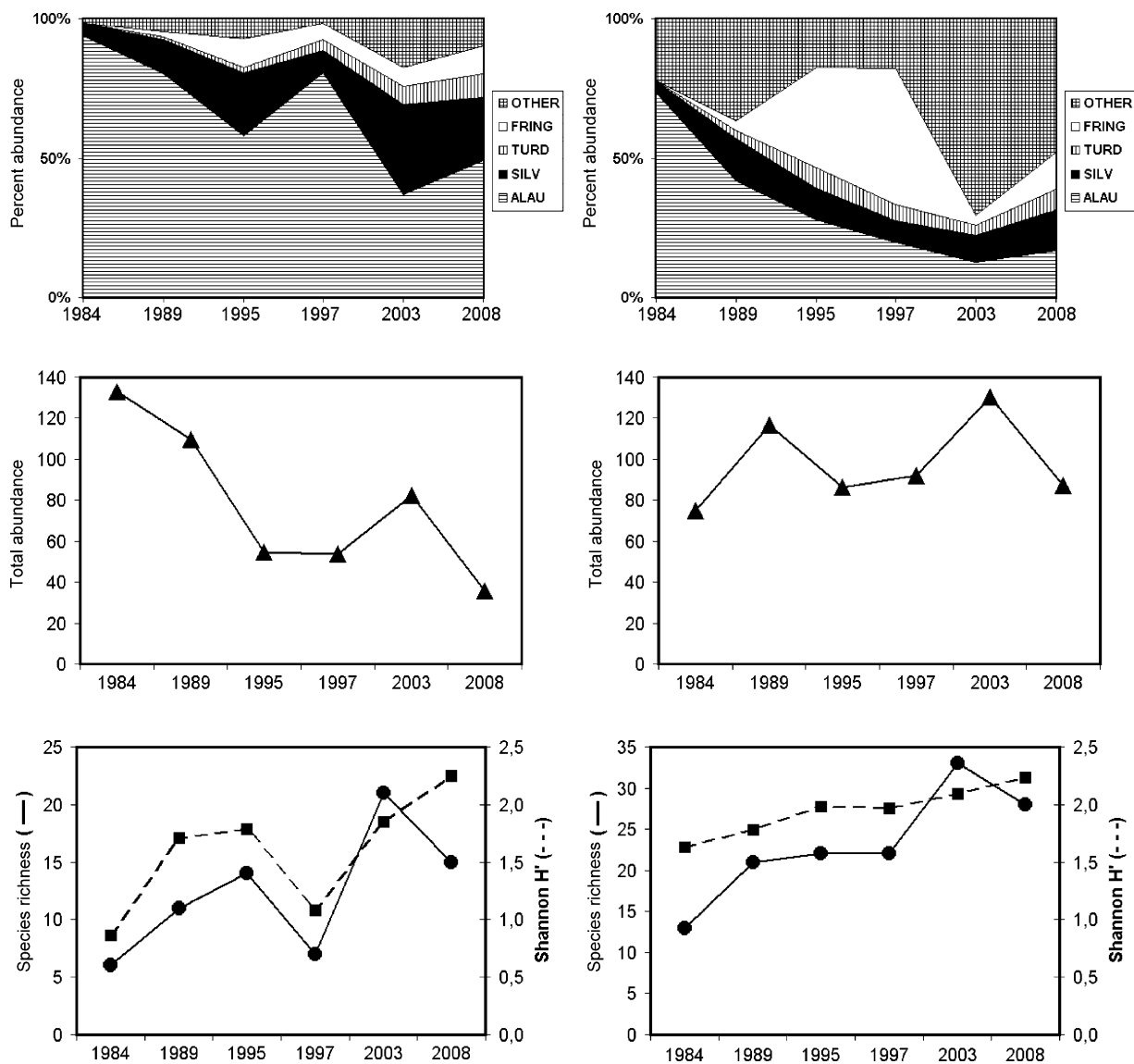


Fig. 3. Changes in the relative abundance of the main families (expressed as % of total IKA), total community abundance and structural parameters of the passerine bird community (left, summer; right, winter).

**Table 2**

Results of GLM showing the response of bird metrics (IKAs of species or families) and indices to habitat variables. Marginally significant ( $0.05 < p < 0.2$ ) but biologically meaningful relationships are also included. Significance levels:  $p < 0.05 = *$ ;  $p < 0.01 = **$ ;  $p < 0.001 = ***$ . Correlation coefficients among variables are also shown.

Bird variables		GLM [sign; adjusted $r^2$ ; significance level; (%explained deviance)]			
		STEP	SALT	REED	CULT
<b>Summer community</b>					
MELC	<i>Melanocorypha calandra</i>	0.99*** (99.34%)	−0.76* (82.47%)		
GALC	<i>Galerida cristata</i>	0.72* (79.52%)	−0.69* (77.48%)		
SYLP	<i>Sylvia conspicillata</i>				0.76* (82.36%)
SATO	<i>Saxicola torquata</i>			0.94** (95.77%)	
CALB	<i>Calandrella brachydactyla</i>				0.80* (85.13%)
MILC	<i>Miliaria calandra</i>				0.76* (82.24%)
CACH	<i>Chloris chloris</i>		0.61ms (70.87%)		
ALAU	Total Alaudidae	0.76* (82.50%)	−0.50ms (50.33%)		
TURD	Total Turdidae	−0.58ms (68.88%)		0.75* (81.40%)	
FRIN	Total Fringillidae	−0.49ms (61.94%)	0.85* (89.17%)		
TOTC	Total community	0.75* (81.40%)	−0.73* (80.20%)		
RDBO	Spanish Red Book Index				0.36* (52.21%)
<b>Winter community</b>					
LAME	<i>Lanius excubitor/meridionalis</i>	−0.93** (95.22%)	0.79* (84.82%)		
CIST	<i>Cisticola juncidis</i>				0.67ms (75.56%)
MELC	<i>Melanocorypha calandra</i>	0.63ms (72.54%)	−0.44ms (58.70%)		
PHOC	<i>Phoenicurus ochruros</i>				0.68ms (76.59%)
EMBS	<i>Emberiza schoeniclus</i>			0.46ms (60.19%)	
STUN	<i>Sturnus unicolor</i>			0.46ms (60.19%)	
PASD	<i>Passer domesticus</i>				−0.38ms (53.88%)
CARD	<i>Carduelis carduelis</i>			0.48ms (61.49%)	
TUME	<i>Turdus merula</i>			0.46ms (60.19%)	
PHYC	<i>Phylloscopus collybita</i>			0.46ms (60.19%)	
ERUB	<i>Erithacus rubecula</i>			0.46ms (60.19%)	
LUSC	<i>Luscinia svecica</i>			0.46ms (60.19%)	
SATO	<i>Saxicola torquata</i>			0.70* (78.09%)	
CALB	<i>Calandrella brachydactyla</i>	0.59ms (69.78%)	−0.58ms (69.00%)		
CALR	<i>Calandrella rufescens</i>		0.34ms (50.62%)		
TURP	<i>Turdus philomelos</i>			0.46ms (60.19%)	
ALAU	Total Alaudidae	0.66ms (74.94%)		−0.48ms (61.7%)	
TURD	Total Turdidae	−0.94** (95.55%)	0.88* (91.02%)		
HSHA	Shannon H'	−0.75** (81.47%)		0.52ms (64.41%)	
BDIR	Birds Directive Index	0.84* (88.73%)	−0.69* (77.44%)		
<b>Pearson correlation coefficients</b>					
STEP	Surface of saline steppe				
SALT	Surface of saltmarsh scrub	−0.92*			
REED	Surface of reedbed	NS	NS		
CULT	Surface of cultures	NS	NS	NS	

community by a saltmarsh (scrubland) one. In the last period (2003–2008) the progressive invasion of the area by reedbeds drives the system succession into more palustrine features.

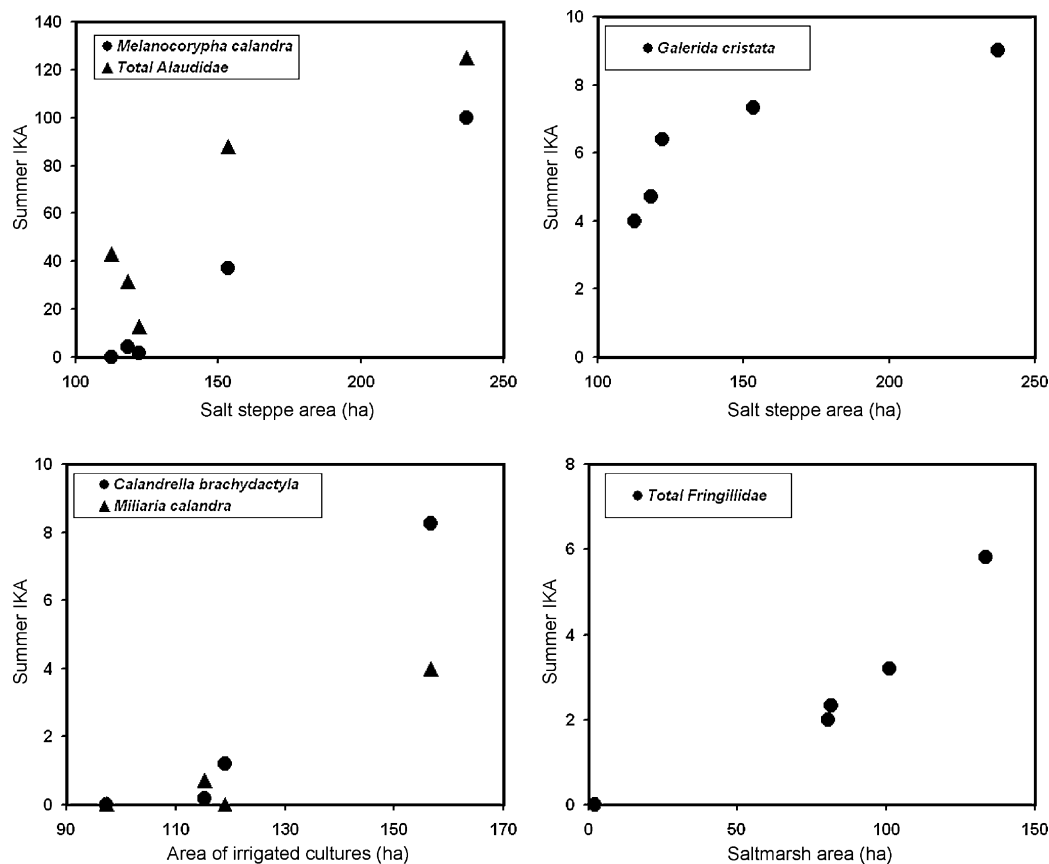
The results of SIMPER analysis highlight the indicator value of some species, and reinforce the interpretation of bird response against changes induced by agricultural practices. In the early period, differences between the summer community of the dryland cultures (1984 samples) and the saline steppe (1989 ones) are marked by lesser short-toed lark (*Calandrella rufescens*), calandra lark (*Melanocorypha calandra*), corn bunting (*Miliaria calandra*) and fan-tailed warbler (*Cisticola juncidis*), dominant in the wetland, while crested lark (*Galerida cristata*), short-toed lark (*Calandrella brachydactyla*), house sparrow (*Passer domesticus*), serin (*Serinus serinus*) and red-legged partridge (*Alectoris rufa*) contribute more on the opposite side (Table 3). In winter some of these differences are attenuated, particularly in *Melanocorypha calandra*, *G. cristata* and *C. rufescens*, as does the general dissimilarity between both habitats.

When we examine the differences between 1989 and 1996 (the central year of the second period), in summer we detect a general decline in the most representative wetland species: *Melanocorypha calandra*, *Miliaria calandra* and *C. juncidis*, but also in *C. rufescens* (Table 4). In winter the situation is much the same, except for *C. rufescens* (Table 5), that increases in the second period, as does the greenfinch (*Carduelis chloris*).

The inclusion of 1995 in the comparisons allows a better interpretation of some changes, mainly the case of *C. rufescens*. In summer, this species peaks in 1989 but is still abundant in 1995, and remains more or less stable thereafter (Table 4). This suggests that the response to saltmarsh expansion occurs earlier in the summer community, since the winter peak appears in 1996 (Table 5).

The last stage (2003 onwards; Table 4) is characterized by a rather heterogeneous community, especially in summer when the species contributing more to the differences are either of the original steppe assemblage (*C. juncidis*, *Melanocorypha calandra*), of urban-agricultural habitats (*P. domesticus*), or of scrubland biotopes (*Saxicola torquata*, *Sylvia melanocephala*). In winter, apart from the important contribution of *Anthus pratensis*, *Motacilla alba* and *Phylloscopus collybita*, the 2003 community retains some of the summer indicator species (*C. juncidis*, *Melanocorypha calandra*, *S. torquata*, *P. domesticus*), while some *Turdidae*, *Fringillidae* and *Sylviidae* already peak in 1995 (Table 5). At the end of the period, wintering species of partial palustrine character like reed bunting (*Emberiza schoeniclus*), as well as some *Turdidae* tend to increase in an apparent response to the partial invasion of saltmarsh by reedbeds.

Among the conservation indices, the SPEC and Spanish Red Data Book based ones show an overall decrease in the summer community (although with a partial recovery in recent years),



**Fig. 4.** Response of some representative species and families of the summer community to habitat variables. *Melanocorypha calandra*, *Galerida cristata*, and total *Alaudidae* are plotted against saline steppe area; *Calandrella brachydactyla* and *Miliaria calandra* against irrigated cultures, and total *Fringillidae* against the area of saltmarsh scrub.

and fluctuant trends with even some gain at the end of the period. The Bird Directive index decline (Fig. 7) is the only significant one (adj.  $r^2 = 0.58$ ;  $p < 0.05$ ) and occurs both in spring and summer. It summarizes the decrease of the Annex I species present, namely *Melanocorypha calandra* and *C. brachydactyla* in summer, and these same species plus *Sylvia undata* in winter. Besides these, the general loss of conservation value in summer is due to the decline of characteristic inhabitants of open spaces and mosaic habitats like *Miliaria calandra* and *Lanius senator* (SPEC3). Among “winner” species adding conservation value we can remark *E. schoenichus* and *Carduelis cannabina* in winter.

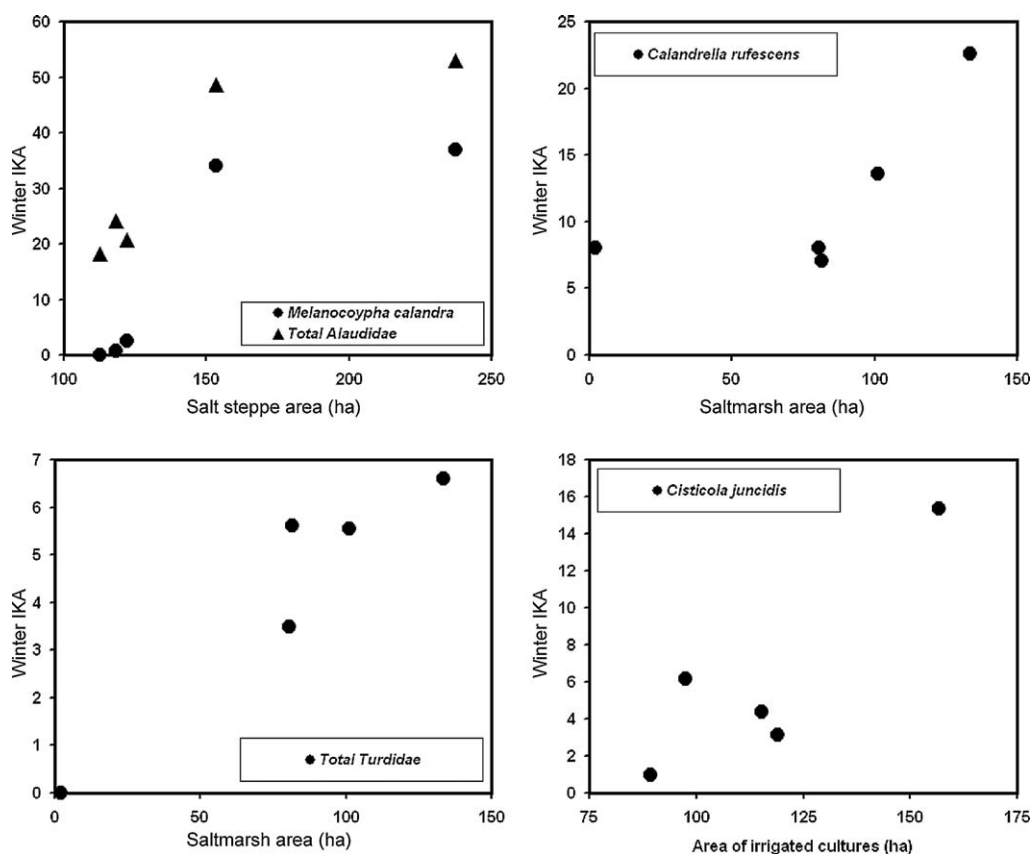
#### 4. Discussion

After Crutzen (2002) coined the term ‘Anthropocene’ to refer to a new geological era in which man has become the driver of global processes, biodiversity patterns cannot be fully explained without considering human impact. This is particularly evident in the Mediterranean Basin, a centre of human influence for millennia (Allen, 2003; Blondel, 2006) and the scenario of a complex coevolution between man and ecosystems (Di Castri et al., 1981). Most Mediterranean ecosystems are so inextricably linked to human interventions that the future of biodiversity cannot be disconnected to that of human affairs (Blondel and Aronson, 1999). In such a context, conservation biologists have explored shortcuts that rely on identifying key species to be focused on during planning efforts directed at preventing, mitigating, and reversing the loss of species, ecosystems, and landscapes (Hess and King, 2002).

Human determinants influence Mediterranean ecosystems along a gradient in which moderate disturbances enhance

biological and landscape diversity, while more intense or prolonged ones cause them to shift across ecological thresholds towards increasingly degraded and exploited systems (Blondel and Aronson, 1999; Brotons and Reunanen, 2005). In this sense, our compilation of bird surveys for a typical semiarid wetland, showed dramatic changes in abundance and community structure, with important implications for the conservation value and function of this ecosystem for birds. The responsiveness of birds to agricultural impacts in wetlands confers them utility in monitoring for conservation, even when the taxon-specific differences in their response to the same sources of disturbance – both among birds and between these and other vertebrate classes –, limits the usefulness of focal taxa as surrogates and as management tools (Lindemayer et al., 2002; Ficetola et al., 2007).

The three phases defined, more evident in the winter plot (Fig. 6), correspond basically with the pattern of irrigated land expansion in the watershed (Carreño et al., 2008), which resembles that of other Mediterranean coastal regions (López and Morales, 2002). In our study area this starts in 1979 and is characterized by sigmoid growth, which begins to slow down in 1991, and finally stabilizes by 2000. The surface of irrigated cultures is highly correlated with the increase of saltmarsh and reedbed (at the expense of salt steppe), a relationship that improves substantially when a 5-year lag is allowed (Carreño et al., 2008). Considering the same time lag, the bird census from the mid and late 1980s would bear the footprint of the early stages of irrigation growth, while those from 1995 to 1997 would represent the peak of this process, and the most recent ones the final stabilization of it. Especially in summer, there is an ample overlap between the last two groups of cases, probably because the loss of saline steppe occurs relatively quickly and is stabilized in the second phase (1997 onwards). From



**Fig. 5.** Response of some representative species and families of the winter community to habitat variables. *Melanocorypha calandra* and total *Alaudidae* are represented against saline steppe area, and total *Turdidae* against the area of saltmarsh scrub and *Cisticola juncidis* against irrigated cultures.

this point there is not a clear dominance of any particular habitat type, and bird community differences probably have to be explained at the scale of microhabitat preferences (Serrano and Astrain, 2005). All these changes are closely associated to the loss of saline steppe in the habitat mosaic, although not all species respond individually. Probably, the lagged response is related to an increased resilience of some species in the first stages of replacement of the saline steppe by saltmarsh, buffered also, by the role of the last patches of dryland crops as supplementary feeding zone. There is also a general trend of winter communities to respond earlier to habitat change.

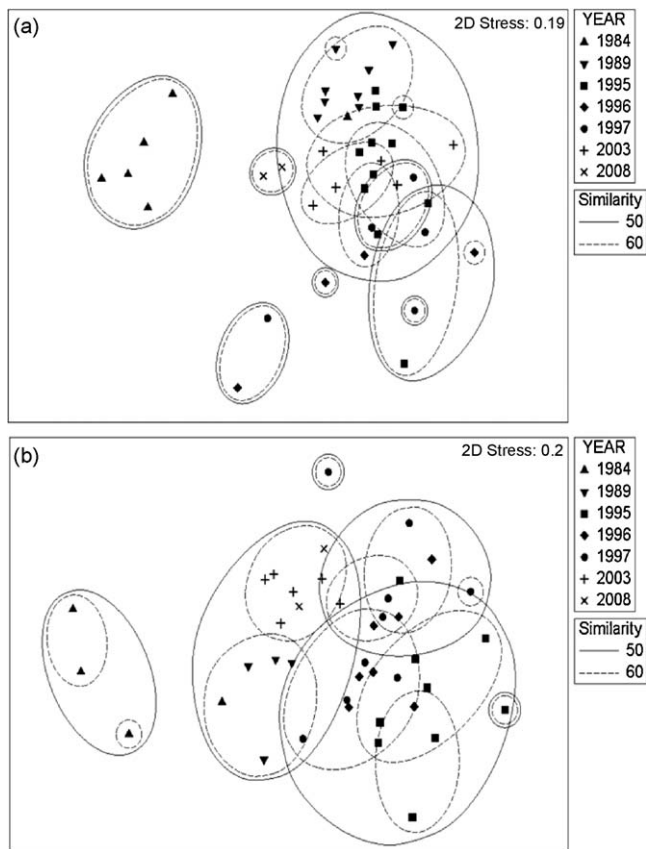
Despite the lack of a clear individual response, changes at the family level characterize the intermediate-late phases (after 1995), when the loss of steppe habitat is stabilized and the development of halophilous scrub reaches its maximum (Carreño et al., 2008). This is the case in both seasons for *Fringillidae*, typical granivores that seem to tolerate the gradual encroachment by scrubland while taking advantage of the increased supply of seeds produced by species like *Sarcocornia* (= *Salicornia*) and *Atriplex*, in addition to *Suaeda* and *Limonium* (Brown and Atkinson, 1996; Kalejta-Summers, 1997; Dierschke and Bairlein, 2004). *Sylviidae*, although favoured in general terms by saltmarsh development, show species-specific responses, probably illustrating different habitat preferences. Spectacled warbler (*Sylvia conspicillata*), present in summer, peaks in the early stages (1989) and disappears towards the end of the 1990s (Table 4). Dartford warbler (*S. undata*), a wintering species, peaks in the early phase, but remains during the whole period of change (Table 5). Sardinian warbler (*S. melanocephala*), the only species present in both seasons, peaks towards the late ones (2003). More palustrine species like reed bunting usually characterize pure reedbeds (Paracuellos, 1996), or transi-

tional (mixed) habitats located between the reed belt of lagoons and the saltmarsh vegetation of immediate terrestrial areas (Peiró, 2006). Other indicator species of this assemblage (*A. pratensis*, *M. alba*, *P. collybita*), typically winter in open scrubland and mixed reedbed mosaics surrounding wetlands (López-Iborra et al., 2005; Peiró, 2006).

*Alaudidae*, the dominant and most genuine family of the steppe assemblage, and to a great extent, of the dryland agricultural community, is the most affected numerically. Three out of four species are positively associated at least in one season with the surface of saline steppe (Figs. 4 and 5). Only the lesser short-toed lark shows an increase during the period of greatest reduction of this habitat (between 1989 and 1997). Since precipitation seems to control general productivity in Mediterranean habitats (Soriquer, 1981), it is reasonable to expect a positive initial effect of increased water availability in the original steppe habitat. De Juana and García (2005) explain population responses of steppe birds on the basis of inter-annual changes in rainfall. The apparent indifference of *C. rufescens* to most habitat variables can be due to the particular preferences of this species in terms of coverage and vertical structure of the vegetation. This lark is common in coastal steppe areas with a good representation of saltmarsh scrub and low vegetation cover (Cramp, 1988; Paracuellos, 1994; Tellería et al., 1999; Torralva et al., 2003). Hernández (1995) added to its preferences, a weaker association with perennial grasses (e.g. *L. spartum*, a dominant species in the saline steppe), and optima at intermediate heights of these two types of vegetation (saltmarsh and steppe).

But the relationship of bird populations and communities with agricultural changes is a complex one (Ursúa et al., 2005). The positive response of *C. brachydactyla* (in summer) to the surface of





**Fig. 6.** Two-dimensional MDS ordination diagram for summer (a) and winter (b) showing the gradient of temporal change (based on bird communities). The 50–60 percentage of similarity level has been used for delineating homogeneous periods. The original resemblance matrix based on Bray–Curtis similarity has been  $\log(x + 1)$  transformed.

cultures is somewhat unexpected since this species usually favours dryland, unploughed agricultural habitats (De Juana and Suárez, 2004), virtually absent from our study area after 1989. It was more abundant in the peripheral dryland cultures surveyed in 1984 (24 birds/km in summer and 14 in winter) than in the Carmolí wetland in any year (8 and 1.78 birds/km as maximum values, respectively). In winter *C. brachydactyla* seems to depend more on the saline steppe habitat. An explanation for the summer preferences is the particular nature of irrigated agriculture in this area. Due to the low quality of water, the consequent salinisation of soils, and the cultivation of short-cycle crops, the fields surrounding the wetland stay uncultivated for long periods, favouring the presence and breeding of many typical steppe species (Fuller et al., 2004). In Murcia Region *C. brachydactyla* is a typical species of uncultivated agricultural areas (Hernández, 1995). The abundance of corn bunting, another species of European concern (Birdlife International, 2004) generally associated with arable land (Diaz and Telleria, 1997; Stoate et al., 2000), is also positively related with the surface of cultures around the wetland. The recent breeding of collared pratincole (*Glareola pratincola*) in these areas, made possible through agreements between landowners and non-governmental conservation organizations (<http://www.asociacionse.org/canastera/>), is an example of their ecological potential. In the future, the conservation of many Mediterranean wetlands will depend critically on the management of the agricultural landscape in which wetlands are embedded (Piorr, 2003; Clergue et al., 2005), especially of the immediate cropland belt that should be also a priority conservation target. The same applies to steppe habitats in agricultural mosaics (e.g. Brotons et al., 2005; Moreira et al., 2005).

**Table 3**

Results of SIMPER (similarity percentages) analysis for the first period of change, showing the main species (differentiating winter and summer community) that determine the dissimilarity between the two first stages: 1984 (initial state reference) and 1989 (first stage of replacement of the saline steppe by saltmarsh).

#### Summer community

1984 vs 1989. Average dissimilarity 66.57

Species	Group 1984		Group 1989			
	Av.Abund	Av.Abund	Av.Diss	Diss/SD	Contrib%	Cum.%
CALR	0.47	3.45	9.54	2.54	14.33	14.33
MELC	1.66	3.52	7.07	1.90	10.62	24.95
MILC	0.00	2.09	6.74	4.97	10.13	35.07
CIST	0.44	2.51	6.71	2.45	10.08	45.16
SERS	2.08	0.00	6.43	1.64	9.66	54.81
CALB	2.66	1.33	6.09	1.34	9.14	63.96
PASD	1.60	0.00	4.85	1.38	7.28	71.24
ALER	1.27	0.00	4.05	1.17	6.09	77.33
GALC	2.66	1.58	3.61	1.09	5.43	82.75

#### Winter community

1984 vs 1989. Average dissimilarity 55.27

Species	Group 1984		Group 1989			
	Av.Abund	Av.Abund	Av.Diss	Diss/SD	Contrib%	Cum.%
CALB	2.45	0.00	7.40	4.27	13.39	13.39
CIST	0.17	2.05	5.44	1.80	9.85	23.24
PASD	1.69	0.00	4.96	1.11	8.98	32.22
CALR	0.55	1.86	4.82	1.87	8.73	40.95
ANTP	2.11	3.61	4.76	1.49	8.62	49.57
CACH	0.00	1.52	4.61	1.61	8.34	57.91
MOTA	1.52	0.68	4.48	1.18	8.11	66.02
SERS	1.17	0.40	3.33	1.20	6.03	72.05
GALC	1.79	1.35	3.08	1.24	5.57	77.62
SYLU	0.27	1.08	2.66	1.43	4.81	82.43
BURO	0.75	0.00	2.36	0.74	4.27	86.70
MELC	3.24	3.47	1.89	1.34	3.42	90.12

It is not possible, nor realistic, to try to explain most variation in all species or groups on the sole basis of the total surface area of the different habitats (Serrano and Astrain, 2005). Nor it can be done solely on the basis of local habitat changes. The size and spatial arrangement of habitat patches, not quantified in the work of Carreño et al. (2008) as well as other sources of habitat heterogeneity, can greatly affect the persistence and/or abundance of species in the community (Benton et al., 2003; Haslem and Bennet, 2008), as do the microhabitat preferences for nesting (Yanes et al., 1996). External factors, illustrated by population trends at higher geographical scales, can also explain the numerical variation, especially of species showing cyclical fluctuations. Unfortunately, the Spanish bird monitoring programs that could illustrate these patterns (Del Moral et al., 2008) cover only the most recent part of our study period.

Using data coming from so many different monitoring schemes, also raises the problem of bias induced by observer, date, sampling effort, and other sources of variation that can be confounded with true changes in population or community parameters (Devictor et al., 2007). But it is quite improbable that all or most of the changes detected simply reflect these methodological factors. Strip transects are considered adequate to track within-site change in the relative abundance of wintering grassland birds, and comparable to area searches in producing density estimates (Roberts and Schnell, 2006). The general concordance of summer and winter patterns, in the species and taxocenoses affected, and in the magnitude of the changes, suggests that sampling bias is not masking the main picture of bird community changes in any season. The habitat variables most related with bird variation (Table 2) also suggest

**Table 4**

Results of SIMPER analysis for the period of transition, showing the main species of summer community that determine the dissimilarity between 1989 (first stage of replacement of the saline steppe by saltmarsh) and the intermediate phases of change (mid 1990s).

Summer community						
First stage: 1989 vs 1996. Average dissimilarity 67.69						
Species	Group 1989	Group 1996				
	Av.Abund	Av.Abund	Av.Diss	Diss/SD	Contrib%	Cum.%
MELC	3.52	0.00	15.40	4.56	22.75	22.75
MILC	2.09	0.00	9.02	4.73	13.32	36.07
CIST	2.51	0.68	8.32	1.94	12.29	48.36
CALR	3.45	1.80	7.94	1.25	11.73	60.10
CALB	1.33	0.00	5.46	0.92	8.07	68.17
GALC	1.58	2.02	3.79	0.96	5.59	73.76
PICV	0.00	0.82	3.67	1.61	5.42	79.18
CACH	0.64	0.27	2.97	0.84	4.39	83.57
STUN	0.00	0.71	2.69	0.57	3.97	87.54
SYLP	0.50	0.00	2.18	0.56	3.22	90.76
Second stage: 1995 vs 1996. Average dissimilarity 55.27						
Species	Group 1995	Group 1996				
	Av.Abund	Av.Abund	Av.Diss	Diss/SD	Contrib%	Cum.%
CIST	2.24	0.68	8.76	1.69	15.07	15.07
CALR	2.93	1.80	8.28	1.06	14.26	29.33
MELC	1.19	0.00	5.55	1.17	9.55	38.88
CACH	1.14	0.27	5.40	1.01	9.29	48.17
PICV	0.10	0.82	4.44	1.19	7.64	55.81
GALC	1.63	2.02	3.94	0.95	6.77	62.58
PASD	0.81	0.00	3.70	1.00	6.36	68.94
STUN	0.10	0.71	3.49	0.61	6.00	74.95
SARU	0.40	0.27	2.40	0.81	4.13	79.08
LANI	0.35	0.27	2.28	0.77	3.93	83.00
SYLM	0.30	0.27	2.14	0.74	3.68	86.68
MILC	0.29	0.00	1.43	0.46	2.46	89.14
SYLP	0.30	0.00	1.43	0.60	2.45	91.59
Last stage: 1996 vs 2003. Average dissimilarity 63.79						
Species	Group 1996	Group 2003				
	Av.Abund	Av.Abund	Av.Diss	Diss/SD	Contrib%	Cum.%
CIST	0.68	2.73	9.29	1.76	14.57	14.57
SATO	0.00	1.53	6.72	2.65	10.54	25.10
MELC	0.00	1.39	6.02	2.60	9.43	34.53
PASD	0.00	1.34	5.71	1.17	8.95	43.49
CALR	1.80	2.19	5.14	1.18	8.06	51.55
GALC	2.02	1.96	4.80	1.42	7.52	59.08
LANI	0.27	1.29	4.60	1.80	7.21	66.29
SYLM	0.27	1.03	3.73	1.68	5.85	72.14
PICV	0.82	0.00	3.68	1.62	5.77	77.91
STUN	0.71	0.31	3.47	0.84	5.45	83.36
CACH	0.27	0.59	2.91	0.83	4.56	87.92
CARD	0.00	0.49	1.92	0.43	3.00	90.92

that summer and winter communities respond in a parallel way to the changes in landscape and vegetation structure driven by irrigation at the watershed scale. Habitat variables, however, are very rough indicators of the compositional and structural features of habitats tied to each species' particular requirements. In any case, the identification of species that contribute most to community change, combined with general trends in indices and habitat relationships, is essential to apply bird monitoring to bioindication.

Although species richness and diversity are often set as general targets in protected area management, the use of more specific conservation indices allows a best evaluation of performance against particular objectives. In our study area, the change towards less steppic conditions represents a small gain in conservation value. The species benefited are usually the

**Table 5**

Results of SIMPER analysis for the period of transition, showing the main wintering species that determine the dissimilarity between 1989 (first stage of replacement of the saline steppe by saltmarsh) and the intermediate phases of change (mid 1990s).

Winter community						
First stage: 1989 vs 1996. Average dissimilarity 51.68						
Species	Group 1989	Group 1996				
	Av.Abund	Av.Abund	Av.Diss	Diss/SD	Contrib%	Cum.%
MELC	3.47	0.16	10.33	4.58	20.00	20.00
CIST	2.05	0.66	5.09	1.58	9.84	29.84
CACH	1.52	2.60	4.28	1.26	8.28	38.12
ANTP	3.61	2.52	4.15	0.93	8.02	46.14
CALR	1.86	3.10	4.12	1.79	7.97	54.12
SATO	0.55	1.65	3.98	1.54	7.69	61.81
GALC	1.35	0.70	3.59	1.22	6.95	68.76
SYLM	0.00	1.09	3.40	2.13	6.57	75.33
SERS	0.40	0.92	2.91	0.84	5.63	80.95
SYLU	1.08	0.63	2.33	1.26	4.52	85.47
Second stage: 1995 vs 1996. Average dissimilarity 47.63						
Species	Group 1995	Group 1996				
	Av.Abund	Av.Abund	Av.Diss	Diss/SD	Contrib%	Cum.%
SARU	1.94	0.16	5.95	2.19	12.48	12.48
SATO	0.00	1.65	5.21	2.09	10.94	23.43
ANTP	1.93	2.52	4.95	1.27	10.38	33.81
CACH	2.67	2.60	4.66	1.40	9.78	43.59
CIST	1.44	0.66	3.70	1.24	7.77	51.36
SERS	0.77	0.92	3.67	1.00	7.71	59.07
SYLU	1.07	0.63	2.71	1.33	5.70	64.77
GALC	0.42	0.70	2.59	0.94	5.44	70.21
PASD	0.85	0.16	2.58	0.88	5.42	75.62
CALR	2.95	3.10	2.51	1.31	5.28	80.90
SYLM	0.84	1.09	2.50	1.27	5.24	86.14
Last stage: 1995 vs 2003. Average dissimilarity 55.48						
Species	Group 1995	Group 2003				
	Av.Abund	Av.Abund	Av.Diss	Diss/SD	Contrib%	Cum.%
ANTP	1.93	4.09	5.35	1.57	9.64	9.64
SATO	0.00	2.03	4.60	3.33	8.29	17.93
PHYC	0.00	1.84	4.00	1.43	7.21	25.14
MOTA	0.00	1.77	3.84	1.92	6.93	32.07
CACH	2.67	1.60	3.61	1.64	6.51	38.58
SARU	1.94	0.48	3.53	2.20	6.36	44.94
GALC	0.42	1.95	3.50	2.02	6.30	51.24
MELC	0.34	1.56	3.15	1.56	5.67	56.92
SYLM	0.84	2.10	2.97	1.47	5.34	62.26
PASD	0.85	1.03	2.83	1.03	5.10	67.36
CIST	1.44	2.56	2.81	1.20	5.07	72.43
CALR	2.95	2.58	2.07	1.44	3.72	76.15
SYLU	1.07	0.67	2.06	1.23	3.72	79.87

less representative of the original conditions and often marginal to the region. There is a debate on the role of man-driven introductions of species compensating local extinctions, particularly in island ecosystems (Sax et al., 2002), although Ficetola and Padoa-Schioppa (2009) demonstrate a net negative effect of humans on species richness in Mediterranean islands. This 'compensatory vs detrimental effect' approach offers a useful framework to evaluate human disturbance in other types of Mediterranean landscapes and habitats, where historical factors can be as important as natural disturbance regimes in explaining landscape heterogeneity (Brotons and Reunanen, 2005). As we have shown, human influence on wetlands can produce some transient positive effect on biodiversity and even a more sustained increase in species richness or alpha diversity. The long-term effect, however, points to a net loss of conservation value which can be related to biotic homogenization (Olden and Poff, 2003).

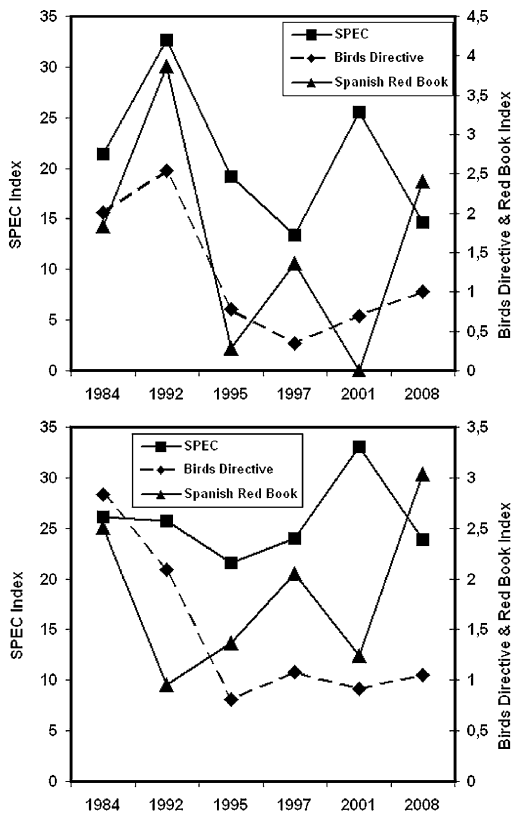


Fig. 7. Changes in the indices used to assess the conservation value of the study area (upper graph, summer; lower graph, winter).

Our ornithological indices (Fig. 7) describe different trajectories depending on season and on the legal or scientific value involved in the calculation: the Birds Directive based index is the only one that shows a marked decline both in summer and winter, while those based on SPEC and IUCN Red Book categories exhibit ups and downs, or even improve along the period of habitat change. But, considering the legal status of the study area, the focus should be put on the former. The Carmolí wetland has been declared as a Bird SPA under 79/409 EC Directive, and the loss in conservation value for Annex I species implies that this function has not been achieved. Among other criteria, this SPA was declared in 2001 on the basis of one steppe passerine bird (*C. rufescens*) exceeding the population threshold established at the European level (Viada, 1998). The effectiveness of protection could be questioned on the basis of its recent population decline. In fact, the ornithological evaluation of the area was made in the period of peak response of *C. rufescens* to increased water input, and after a main change in the original physiognomy. Our estimates of summer population density for this species are 20 birds/10 ha at the start of the study, rising to 44 and 47.5 birds/10 ha in 1989 and 1997, which probably represent the highest densities of the Iberian Peninsula (Hernández and Pela, 1987; Suárez et al., 2002; Sampietro and Pelayo, 2003). This transient positive effect can be attributed, as we have suggested, to the expansion of saltmarsh vegetation. In the long term, however, its abundance is lowered to values between 5.5 and 8.75 birds/10 ha, still high but nearly an order of magnitude lower than the peak values (on which the SPA evaluation is based). Despite its relative tolerance to changes in saltmarsh structure, Serrano and Astrain (2005) consider *C. rufescens* a more specialized bird than its congeneric *brachydactyla*. In the long term this poses a greater threat on the first species, unable to take advantage of alternative habitats (e.g. cropland).

## 5. Synthesis and conclusions

The surface of natural steppe habitat has been reduced in many European areas, due to the development of irrigation (Suárez et al., 1997; Wolff et al., 2001; Brotons et al., 2004; Ursúa et al., 2005; Laiolo and Tella, 2006), leading to declines in their bird populations. The indirect impact of irrigation on coastal steppic areas is an equally important threat to bird populations and communities, as we have seen in the Carmolí wetland. This area exemplifies the deterioration of a steppe area through hydrological changes in the watershed, without direct habitat reclamation. The value of terrestrial birds as indicators of agricultural intensification is illustrated by their integrative, multi-stage response to the habitat changes induced by irrigation. Although monitoring and evaluation tools have been developed to assess the impacts of landscape changes on bird species of conservation concern (Pons et al., 2003), few quantitative studies have applied these tools to Mediterranean steppe systems (Brotons et al., 2004).

Waterbird species (e.g. grebes, cormorants and coot), have shown to be a valuable tool in the surveillance of agricultural and urban water discharge and nutrient loading into wetlands and coastal waters (Roomen et al., 2006), as they do in the Mediterranean (Esteve et al., 2008; Robledano et al., 2006). These taxa, however, have lower conservation scores (none of the species of this guild present in the Mar Menor lagoon qualifies for SPA designation). The warning role of waterbirds with respect to nutrient loading in the Mar Menor lagoon (suggested as early as mid 1980s), could also have been played by steppe passerines with regard to habitat changes in the peripheral wetlands (evident by mid 1990s, and dramatic at the start of this century), had their populations been monitored more continuously. Multispecies assessments of protected areas are a valuable tool, not only for its own management, but for the investigation of human impacts on biodiversity at a wide array of scales (Devictor et al., 2007).

The original landscape and habitat setting of a hypersaline, oligotrophic lake fringed by steppe habitats has been gradually modified towards the present scenario of a Mediterranean-like water mass with incipient eutrophication symptoms, surrounded by expanding saltmarsh scrub and reedbed communities. These changes seem to be threatening the typical aquatic and terrestrial biota. The need to manage agricultural impacts at the scale of Mediterranean wetlands' watersheds (Álvarez-Cobelas et al., 2005) emerges here as a critical biodiversity issue. The trends we have observed question the effectiveness of the protection measures applied at the scale of wetlands, and will continue in the future unless action is taken in their whole area of influence.

## Acknowledgements

Vicente Hernández Gil kindly provided the results of passerine bird surveys carried out within his Ph.D. research in the Marina del Carmolí. Passerine data for 1995–1997 were gathered as a part of the wetland monitoring scheme financed by Project LIFE/1973/93/11-10: "Conservación de humedales y otros ecosistemas característicos de zonas áridas". Other research on which this paper is based has been financed by the European Union through EC "Energy, Environment & Sustainable Development Programme" (DITTY Project, Contract EVK3-CT-2002-0084 DITTY) and by the Spanish Ministry of Education and Science through Project IBISMED-Estado ecológico de los humedales del Mediterráneo semiárido: propuesta de indicadores para su evaluación (CGL 2006-08134). Preliminary results of this research project have been included in a synthesis chapter in the book edited by Russo, R.E. (in press): "Wetlands: Ecology, Conservation & Restoration", Nova Science Publishers, New York. The comments of an

anonymous reviewer helped improve an earlier version of the manuscript.

## References

- Allen, H., 2003. Response of past and present Mediterranean ecosystems to environmental change. *Prog. Phys. Geog.* 27 (3), 359–377.
- Álvarez-Cobelas, M., Rojo, C., Angeler, D., 2005. Mediterranean limnology: current status, gaps and the future. *J. Limnol.* 64 (1), 13–29.
- Álvarez-Rogel, J., Jiménez-Cárceles, F.J., Nicolás, C., 2006. Phosphorus and nitrogen content in the water of a coastal wetland in the Mar Menor Lagoon (SE Spain): relationships with effluents from urban and agricultural areas. *Water Air Soil Poll.* 173 (1), 21–38.
- Álvarez-Rogel, J., Jiménez-Cárceles, F.J., Roca, M.J., Ortiz, R., 2007. Changes in soils and vegetation in a Mediterranean coastal salt marsh impacted by human activities. *Estuar. Coast. Shelf Sci.* 73 (3–4), 510–526.
- Benton, T.G., Vickery, J.A., Wilson, J.D., 2003. Farmland biodiversity: is habitat heterogeneity the key? *Trends Ecol. Evol.* 18 (4), 182–188.
- Birdlife International, 2004. Birds in Europe. Population Estimates, Trends and Conservation Status. Birdlife Conservation Series 12, Cambridge.
- Blondel, J., 2006. The 'design' of Mediterranean landscapes: a millennial story of humans and ecological systems during the historic period. *Hum. Ecol.* 34, 713–729.
- Blondel, J., Aronson, J., 1999. *Biology and Wildlife of the Mediterranean Region*. Oxford University Press, New York.
- Brotos, L., Mañosa, S., Estrada, J., 2004. Modelling the effects of irrigation on the distribution of steppe birds in Mediterranean farmland. *Biodivers. Conserv.* 13, 1039–1058.
- Brotos, L., Reunanen, P., 2005. The role of landscape patterns and land-use history in the composition of bird assemblages in Mediterranean and boreal forest ecosystems. *Revista Catalana d'Ornitologia* 21, 47–57.
- Brotos, L., Wolff, A., Paulus, G., Martin, J., 2005. Effect of adjacent agricultural habitat on the distribution of passerines in natural grasslands. *Biol. Conserv.* 124 (3), 407–414.
- Browder, S.F., Johnson, D.H., Ball, I.J., 2002. Assemblages of breeding birds as indicators of grassland condition. *Ecol. Indic.* 2 (3), 257–270.
- Brown, A.F., Atkinson, P.W., 1996. Habitat associations of coastal wintering passerines. *Bird Study* 43 (2), 188–200.
- Burfield, I., 2005. The conservation status of steppe birds in Europe. In: Bota, G., Morales, M.B., Mañosa, S., Camprodon, J. (Eds.), *Ecology and Conservation of Steppe-land Birds*. Lynx Edicions, Barcelona, pp. 119–140.
- Caro, T.M., O'Doherty, G., 1999. On the use of surrogate species in conservation biology. *Conserv. Biol.* 13 (4), 805–814.
- Carreño, M.F., Esteve, M.A., Martínez, J., Palazón, J.A., Pardo, M.T., 2008. Habitat changes in coastal wetlands associated to hydrological changes in the watershed. *Estuar. Coast. Shelf Sci.* 77, 475–483.
- Clergue, B., Amiaud, A., Pervanchon, F., Lasserre-Joulin, F., Plantureux, S., 2005. Biodiversity: function and assessment in agricultural areas. A review. *Agron. Sustain. Dev.* 25, 1–15.
- Cramp, S. (Ed.), 1988. *The Birds of the Western Palearctic*, Vol. 5: Tyrant Flycatchers to Thrushes. Oxford University Press, London.
- Crutzen, P.J., 2002. Geology of mankind. *Nature* 415, 23.
- De Juana, E., Suárez, F., 2004. Terrera Común *Calandrella brachydactyla*. In: Madroño, A., González, C., Atienza, J.C. (Eds.), *Libro Rojo de las Aves de España*. Dirección General para la Biodiversidad-SEO/Birdlife, Madrid, pp. 312–314.
- De Juana, E., García, A., 2005. Fluctuaciones relacionadas con la precipitación en la riqueza y abundancia de aves de medios esteparios mediterráneos. *Ardeola* 52 (1), 53–66.
- Del Moral, J.C., Escandell, V., Molina, B., Bermejo, A., Palomino, D. (Eds.), 2008. *Programas de seguimiento de SEO/Birdlife en 2006*. SEO/Birdlife, Madrid.
- Devictor, V., Godet, L., Julliard, R., Couvet, D., Jiguet, F., 2007. Can common species benefit from protected areas? *Biol. Conserv.* 139 (1–2), 29–36.
- Di Castri, F., Goodall, W., Specht, R.L., 1981. Mediterranean-type Shrublands. *Ecosystems of the World*, vol. II. Elsevier, Amsterdam.
- Diaz, M., Telleria, J.L., 1997. Habitat selection and distribution trends of corn buntings in the Iberian Peninsula. In: Donald, P., Aebischer, N.J. (Eds.), *The Ecology and Conservation of Corn Buntings *Miliaria calandra**. JNCC, Peterborough, pp. 151–161.
- Dierschke, J., Bairlein, F., 2004. Habitat selection of wintering passerines in salt marshes of the German Wadden Sea. *J. Ornithol.* 145, 48–58.
- Duelli, P., Obrist, M.K., 2003. Biodiversity indicators: the choice of values and measures. *Agric. Ecosyst. Environ.* 98 (1–3), 87–98.
- Esteve, M.A., Carreño, M.F., Robledano, F., Martínez-Fernández, J., Miñano, J., 2008. Dynamics of coastal wetlands and land use changes in the watershed: implications for the biodiversity. In: Russo, R.E. (Ed.), *Wetlands: Ecology, Conservation & Restoration*. Nova Science Publishers, New York.
- Everard, M., 2008. Selection of taxa as indicators of river and freshwater wetland quality in the UK. *Aquat. Conserv.* 18, 1052–1061.
- Ficetola, G.F., Sacchi, R., Scalic, S., Gentili, A., De Bernardi, F., Galeotti, P., 2007. Vertebrates respond differently to human disturbance: implications for the use of a focal species approach. *Acta Oecol.* 31 (1), 109–118.
- Ficetola, G.F., Padoa-Schioppa, E., 2009. Human activities alter biogeographical patterns of reptiles on Mediterranean islands. *Global Ecol. Biogeogr.* 18 (2), 214–222.
- Fuller, R.J., Hinsley, S.A., Swetnam, R.D., 2004. The relevance of non-farmland habitats, uncropped areas and habitat diversity to the conservation of farmland birds. *Ibis* 146 (Suppl. 2), 22–31.
- Gibbs, J.P., 2000. Wetland loss and biodiversity conservation. *Conserv. Biol.* 14 (1), 314–317.
- Gregory, R.D., van Strien, A., Vorisek, P., Gmelig Meyling, A.W., Noble, D.G., Foppen, R.P.B., Gibbons, D.W., 2005. Developing indicators for European birds. *Philos. T. Roy. Soc. B* 360, 269–288.
- Guirao, A., 1859. Catálogo Metódico de las Aves observadas en una gran parte de la provincia de Murcia. *Boletín de la Real Academia de Ciencias de Madrid* 4, 1–50.
- Haslem, A., Bennet, A.F., 2008. Birds in agricultural mosaics: the influence of landscape pattern and countryside heterogeneity. *Ecol. Appl.* 18 (1), 185–196.
- Hernández, V., 1995. *Ecología de las estepas de la Región de Murcia*. Estructura y dinámica de sus comunidades orníticas. Cuadernos de Ecología y Medio Ambiente, 4, Universidad de Murcia.
- Hernández, F., Pela, E., 1987. Sobre comunidades de aves esteparias en planicies del valle medio del Ebro. I. Int. Simp. Aves Esteparias, León (Spain), pp. 379–393.
- Hess, G.R., King, T.J., 2002. Planning open spaces for wildlife. I. Selecting focal species using a Delphi survey approach. *Landscape Urban Plan.* 58 (1), 25–40.
- Jansen, A., Robertson, A.I., 2001. Riparian bird communities in relation to land management practices in floodplain woodlands of south-eastern Australia. *Biol. Conserv.* 100 (2), 173–185.
- Jiménez-Cárceles, F.J., Álvarez-Rogel, J., 2008. Phosphorus fractionation and distribution in salt marsh soils affected by mine waste and eutrophicated water: a case study in SE Spain. *Geoderma* 144 (1–2), 299–309.
- Kalejta-Summers, B., 1997. Diet and habitat preferences of wintering passerines on the Taff/Ely saltmarshes. *Bird Study* 44, 367–373.
- Laiolo, P., Tella, J.L., 2006. Fate of unproductive and unattractive habitats: recent changes in Iberian steppes and their effects on endangered avifauna. *Environ. Conserv.* 33 (3), 223–232.
- Lindemayer, D.B., Manning, A.D., Smith, P.L., Possingham, H.P., Fischer, J., Oliver, I., McCarthy, M.A., 2002. The focal species approach and landscape restoration: a critique. *Conserv. Biol.* 16 (2), 338–345.
- López, A., Morales, C., 2002. Integrated management of river basins in the Mediterranean region: sustainable water for nature and food. *UICN Mediterranean Office*, Málaga.
- López-Iborra, G., Limiñana, R., Peñarrubia, S.G., Pinheiro, R.T., 2005. Diet of common chiffchaffs *Phylloscopus collybita* wintering in a wetland in south-east Spain. *Revista Catalana d'Ornitologia* 21, 29–36.
- Madroño, A., González, C., Atienza, J.C. (Eds.), 2004. *Libro Rojo de las Aves de España*. Dirección General para la Biodiversidad-SEO/Birdlife, Madrid.
- Mander, L., Cutts, N.D., Allen, J., Mazik, K., 2007. Assessing the development of newly created habitat for wintering estuarine birds. *Estuar. Coast. Shelf Sci.* 75 (1–2), 163–174.
- Maltby, E., 1986. *Waterlogged Wealth*. Earthscan, London.
- Michelson, D.B., Liljeberg, B.M., Pilesjö, P., 2000. Comparison of algorithms for classifying Swedish landcover using Landsat TM and ERS-1 SAR data. *Remote Sens. Environ.* 71, 1–15.
- Moreira, F., Beja, P., Morgado, R., Reino, L., Gordinho, L., Delgado, A., Borralho, R., 2005. Effects of field management and landscape context on grassland wintering birds in Southern Portugal. *Agric. Ecosyst. Environ.* 109 (1–2), 59–74.
- Moreno, D., Pedrocchi, C., Comín, F.A., García, M., Cabezas, A., 2007. Creating wetlands for the improvement of water quality and landscape restoration in semi-arid zones degraded by intensive agricultural use. *Ecol. Eng.* 30 (2), 103–111.
- Olden, J.D., Poff, N.L., 2003. Toward a mechanistic understanding and prediction of biotic homogenization. *Am. Nat.* 162 (4), 442–460.
- Padoa-Schioppa, E., Baietto, M., Massa, R., Bottoni, L., 2006. Bird communities as bioindicators: the focal species concept in agricultural landscapes. *Ecol. Indic.* 6, 83–93.
- Paquet, J.-Y., Vandevyvre, X., Delahaye, L., Rondeux, J., 2006. Bird assemblages in a mixed woodland-farmland landscape: the conservation value of silviculture-dependent open areas in plantation forest. *Forest Ecol. Manage.* 227 (1–2), 59–70.
- Paracuellos, M., 1994. Dinámica anual de la comunidad de Passeriformes en un saladar litoral del sudeste ibérico. *Doñana, Acta Vertebrata* 21, 119–130.
- Paracuellos, M., 1996. Dinámica anual de la comunidad de Passeriformes en carrizales costeros del sudeste ibérico. *Doñana, Acta Vertebrata* 23, 33–44.
- Pardo, M.T., Esteve, M.A., Giménez, A., Martínez-Fernández, J., Carreño, M.F., Serrano, J., Miñano, J., 2008. Assessment of hydrological alterations on wandering beetle assemblages (coleoptera: Carabidae and Tenebrionidae) in coastal wetlands of arid Mediterranean systems. *J. Arid Environ.* 72 (10), 1803–1810.
- Pons, P., Lambert, B., Rigolot, E., Prodon, R., 2003. The effects of grassland management using fire on habitat occupancy and conservation of birds in a mosaic landscape. *Biodivers. Conserv.* 12 (9), 1843–1860.
- Peiró, I.G., 2006. Estudios ornitológicos aplicados a la gestión del hábitat de los Passeriformes del carrizal en el Parque Natural de El Fondo (Alicante, SE de España). Tesis Doctoral. Universidad de Murcia.
- Pierr, H.-P., 2003. Environmental policy, agri-environmental indicators and landscape indicators. *Agric. Ecosyst. Environ.* 98, 17–33.
- Richards, J.A., 1995. *Remote Sensing Digital Image Analysis, An Introduction*, second ed. Springer, Berlin.
- Roberts, J.P., Schnell, G.D., 2006. Comparison of survey methods for wintering grassland birds. *J. Field Ornithol.* 77 (1), 46–60.
- Robledano, F., Esteve, M.A., Hernández Gil, V., Blanco Gago, J.C., Pardo, M.T., Martínez, J., Carreño, M.F., 2006. Cambios en la actividad agrícola y respuesta de las comunidades de passeriformes en los humedales del Mar Menor (Murcia, SE España). In: *Congreso Ibérico de Ornitología*, Elche (Alicante, Spain).

- Roomen, M. van, Koffijberg, K., Noordhuis, R., Soldaat, L., 2006. Long-term waterbird monitoring in The Netherlands: a tool for policy and management. In: Boere, G.C., Galbraith, C.A., Stroud, D.A. (Eds.), *Waterbirds Around the World*. The Stationery Office, Edinburgh, UK, pp. 463–470.
- Sampietro, F.J., Pelayo, E., 2003. Terrera marismeña *Calandrella rufescens*. In: Martí, R., del Moral, J.C. (Eds.), *Atlas de las Aves reproductoras de España*. Dirección General de Conservación de la Naturaleza-Sociedad Española de Ornitología, Madrid, pp. 370–371.
- Santos, T., Suárez, F., 2005. Biogeography and population trends of steppe land birds. In: Bota, G., Morales, M.B., Mañosa, S., Camprodon, J. (Eds.), *Ecology and Conservation of Steppe-land Birds*. Lynx Edicions, Barcelona, pp. 69–102.
- Sax, D.F., Gaines, S.D., Brown, J.H., 2002. Species invasions exceed extinctions on islands worldwide: a comparative study of plants and birds. *Am. Nat.* 160, 766–783.
- Serrano, D., Astrain, C., 2005. Microhabitat use and segregation of two sibling species of *Calandrella* larks during the breeding season: conservation and management strategies. *Biol. Conserv.* 125 (3), 391–397.
- Soriguer, R.C., 1981. Biología y dinámica de una población de conejos (*Oryctolagus cuniculus* L.) en Andalucía occidental. Doñana, *Acta Vertebrata* 8, 1–379.
- Stoate, C., Borralho, R., Araújo, M., 2000. Factors affecting corn bunting *Miliaria calandra* abundance in a Portuguese agricultural landscape. *Agric. Ecosyst. Environ.* 77 (3), 219–226.
- Suárez, F., Naveso, M.A., De Juana, E., 1997. Farming in the drylands of Spain: birds of the pseudosteppes. In: Pain, D., Pienkowski, M.W. (Eds.), *Farming and Birds in Europe*. Academic Press, London, pp. 297–330.
- Suárez, F., Garza, V., Morales, M.B., 2002. Habitat use of two sibling species, the short-toed *Calandrella brachydactyla* and the lesser short-toed *C. rufescens* Larks, in mainland Spain. *Ardeola* 49 (2), 259–272.
- Tellería, J.L., Asensio, B., Díaz, M., 1999. *Aves ibéricas, Volumen II. Passeriformes*. José Manuel Reyero, Madrid.
- Torralva, M., Vargas, M., Jiménez, R., Arcas, J.M., García, A., Robles, M., Caballero, A., 2003. Análisis de la viabilidad de la comunidad ornitológica. In: *Estudio de la Fauna del Mar Menor y directrices para su manejo y conservación*. Vol. II. Technical Document. Consejería de Agricultura, Agua y Medio Ambiente, Comunidad Autónoma de la Región de Murcia.
- Tucker, G.M., 1997. Priorities for bird conservation in Europe: the importance of the farmed landscape. In: Pain, D.J., Pienkowski, M.W. (Eds.), *Farming and Birds in Europe: The Common Agricultural Policy and its Implications for Birds Conservation*. Academic Press, London, pp. 79–116.
- Ursúa, E., Serrano, D., Tella, J.L., 2005. Does land irrigation actually reduce foraging habitat for breeding lesser kestrels? The role of crop types. *Biol. Conserv.* 122, 643–648.
- Viada, C. (Ed.), 1998. *Áreas Importantes para las Aves en España*, 2nd ed. Monografía nº 5, SEO/Birdlife, Madrid.
- Vidal-Abarca, M.R., Esteve Selma, M.A., Suárez Alonso, M.L. (Eds.), 2003. *Los Humedales de la Región de Murcia: Humedales y Ramblas de la Región de Murcia*. Consejería de Agricultura, Agua y Medio Ambiente, Murcia.
- Wolff, A., Paul, J., Martin, J., Bretagnolle, V., 2001. The benefits of extensive agriculture to birds: the case of the little bustard. *J. Appl. Ecol.* 38 (5), 963–975.
- Yanes, M., Herranz, J., Suárez, F., 1996. Nest microhabitat selection in larks from a European semi-arid shrub-steppe: the role of sunlight and predation. *J. Arid Environ.* 32, 469–478.
- Zamorano Ruiz, M., 1932. *Notas sobre algunas especies de los antiguos órdenes de Palmípedas y Zancudas, que frecuentan el Mar Menor (Murcia)*. Librería Melero, Cartagena.