

*Chapter 4*

**DYNAMICS OF COASTAL WETLANDS AND LAND USE  
CHANGES IN THE WATERSHED:  
IMPLICATIONS FOR THE BIODIVERSITY**

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**ABSTRACT**

The Mediterranean coastal landscapes have suffered significant changes along the last decades due to the agricultural intensification and tourist development. Such changes have modified the water flows and specifically the hydrological regime of wetlands, as has occurred in the Mar Menor (Southeast Spain). The Mar Menor coastal lagoon and associated wetlands present noticeable ecological and biodiversity values. However, the land-use changes in the watershed and the consequent changes in the water and nutrient flows along the period 1980-2005 are threatening the conservation of these wetlands. A dynamic model been developed to simulate the key environmental and socio-economic factors driving the export of nutrients to the Mar Menor lagoon and associated wetlands, where some eutrophication processes have appeared.

In the present chapter the changes in the vegetal and faunistic assemblages are analysed. Vegetal communities are studied by means of remote sensing techniques, which have provided information about the changes in area and habitat composition of the wetlands along the considered period. This has shown that the habitats more negatively affected by the hydrological changes are those most threatened in the international context and with a highest interest from the point of view of biodiversity conservation. It has also been possible to verify the direct relationships between all these changes at wetlands scale and the agricultural changes at the watershed scale.

Two faunistic communities especially sensitive to these ecosystemic changes have also been studied: i) Wandering beetles and ii) Birds (waterbirds and steppe passerines). Wandering beetles (Coleoptera) were studied with pitfall traps in 1984, 1992 and 2003 and steppe passeriforms with line transects in several years along the period. In both communities evident changes have been observed. Regarding beetles, the most

halophilous species have been favoured, some of them especially relevant due to its rarity in the European context. The ratio Carabidae/Tenebrionidae has shown to be a good indicator of the hydrological changes of the wetlands. Waterbirds have shown dramatic changes in their relative abundances within the lagoon, with a long-term decline in the most characteristic original species, increases in generalist piscivores and a recent appearance and rapid growth of the herbivores guild. In the case of steppe passeriforms, this community has been negatively affected, especially some species like *Melanocorypha calandra*. The family *Alaudidae* has lost importance to the benefit of the families *Turdidae* and *Fringillidae*. These changes can be considered a loss of value in relation with the original passeriform community, since the wetland qualifies as a Specially Protected Area under the EU's Bird Directive, precisely on the basis of its genuine steppe bird assemblage.

In conclusion the changes at wetlands scale clearly reflect the hydrological modifications at the watershed scale and have significant effects on the most characteristic biodiversity of the wetlands of coastal arid systems.

## INTRODUCTION

The importance of wetlands is increasingly recognised as systems supporting an specific and valuable biodiversity and as areas playing a key role in essential ecological functions such as the control of nutrient flows and the removal of diffuse pollution at the landscape scale. This has promoted different protection and conservation strategies which in the context of arid landscapes, as those existing in Southeastern Spain have an especial importance due to the singularity and key role of wetlands in such arid systems. 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 on the protection and conservation goals. Land use and management practices at watershed scale affect the wetlands in many ways and one of the most relevant ones is linked to the water and nutrients flows entering into the wetlands from the watershed. To what extent do wetlands react to land use changes in the watershed?; Can these changes be tracked in different ecological compartments of the wetlands?; Do that changes draw a similar pattern?. We have tried to answer some of such questions studying a complex of wetlands in a arid landscape which have suffered important long-term land use changes: The Mar Menor lagoon and associated wetlands.

The Mar Menor lagoon is a hypersaline Mediterranean coastal lagoon located in Southeast Spain. Ramsar Site since 1994, it 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 width varying between 100 and 1200 m., with a very narrow connection with the open sea.

Associated to its internal shore there is a series of coastal wetlands, Marina del Carmolí, Playa de la Hita and Saladar de lo Poyo (figure 1), which are protected at both national and international level (Ramsar site and Site of Community Importance for the 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. They are defined as coastal crypto-wetlands (Vidal-Abarca et al. 2003).



Figure 1. Location of Mar Menor wetlands. PH: Playa de la Hita; MC: Marina del Carmolí; LP: Saladar de Lo Poyo.

The Mar Menor watershed is a 1,200 km<sup>2</sup> plain slightly inclined towards the lagoon and drained by several ephemeral watercourses (ramblas), which flow into the lagoon after episodic storm rainfall events. 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. Agriculture is the predominant land-use in the Mar Menor watershed.

Urban changes, tourist development and especially the spread of irrigated lands favoured by the opening of the Tagus-Segura water transfer system in 1979, have led to a significant increase in the water and nutrient flows reaching the Mar Menor lagoon-associated wetlands complex. In the wetlands this has caused changes which are relevant at watershed scale, due to the role of the active wetland area in the removal of nutrients from diffuse sources, and also at wetland scale, 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.

Previous studies on Mar Menor wetlands have focused on conservation and management (Robledano et al., 1987, 1991a, 1991b, Robledano and Esteve, 1992; Ortega et al., 1992), on restoration (Robledano, 1995) and on specific topics such as heavy metals in Lo Poyo wetland (Álvarez-Rogel et al., 2002b, 2004) and nutrients (Álvarez-Rogel et al., 2002a, 2006, 2007a).

In the present chapter the changes in the vegetal and faunistic assemblages of Mar Menor wetlands are studied to analyse their responses to the land use and hydrological changes at watershed scale, the implications in terms of biodiversity and conservation value and to assess potential indicators in the analysed assemblages of the long-term watershed changes. It has been studied the vegetal communities in the Mar Menor wetlands and two faunistic communities especially sensitive to the ecosystemic changes: wandering beetles and birds (waterbirds and steppe passerines). These communities have been studied at specific spatial scales and study areas: vegetal communities were studied in three wetlands associated to the Mar Menor shore (Playa de la Hita, Marina del Carmolí and Saladar de Lo Poyo); wandering beetles and steppe passerines were analysed in the Marina del Carmolí wetland whereas waterbirds were surveyed in the whole Mar Menor lagoon.

## LAND USE CHANGES AND THEIR EFFECTS ON HABITATS OF COASTAL WETLANDS

### Study Area and Methodology

Three Mar Menor wetlands, Playa de la Hita, Marina del Carmolí and Saladar de Lo Poyo, have been studied to analyse the temporal and spatial changes in the area and in the internal composition of such wetlands from 1984 to 2001, by means of remote sensing, in order to analyse their implications, especially as regards the application of the Habitats Directive (92/43/CEE).

All studied wetlands present salt steppes, salt marshes, reedbeds and sandy areas, although with a different relative importance. Following the typology of the Habitats Directive (92/43/CEE), the sandy areas unit is mostly composed of the following habitats: 1210 "Annual vegetation of drift lines" and 2210 "Fixed beach dunes with *Crucianellion maritima*". The salt steppe unit is 95% composed of the priority habitat 1510 "Mediterranean salt steppes, *Limonieta*", whereas the remaining 5% comprises habitat 1430 (Halophilous 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 salt marsh 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 salt marsh are *Sarcocornia fruticosa*, *Arthrocnemum macrostachyum*, *Halimione portulacoides* and *Limonium cossonianum*. Finally the reedbeds unit is dominated by *Phragmites australis*. All habitats are designated 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.

Remote sensing has been extensively used in wetlands studies and inventories (Noriega and Lozano-García, 2000), in the detection of water bodies and vegetation in wetlands (Toyrä et al. 2001), in the elaboration of land use-land cover and vegetation maps (Cihlar et al., 1996; Michelson et al., 2000; Hess et al., 2003; Wang y Tenhunen, 2004; Kumar Joshi et al., 2006), including those of wetlands (Wang et al., 2007), in the detection of land use-land cover

changes (Narumalani *et al.*, 2004; Cakir *et al.*, 2006; Mundia and Aniya, 2006) and in the analysis of hydrological and land cover changes in wetlands (McHugh *et al.*, 2007).

Land cover maps for the three 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 five years of the period 1984 to 2001 (years 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 and Playa de la Hita and 89% in Saladar de Lo Poyo).

The following land cover classes were identified (Carreño *et al.*, 2008): natural vegetation (three units: salt steppe, salt marsh and reedbeds); agricultural fields, 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 the tracking of the changes in the area and in the internal composition of wetlands between 1984 and 2001. Data were analysed under the statistical package R (The R Foundation for Statistical Computing, 2005, <http://www.r-project.org>).

Field data on 2003 and 2004 of soil moisture and conductivity in 35 sample units of the whole set of Mar Menor wetlands were also available and were used in this work.

## Results and Discussion

### *Playa de la Hita Wetland*

Figure 2 shows the land cover maps of Playa de la Hita obtained by supervised classification of Landsat TM+ images between 1984 and 2001. In Playa de la Hita there was a slight reduction in the area of natural vegetation due to the construction of infrastructures (table 1). Regarding its internal composition, the main changes among habitats are the conversion of 29% of salt-steppe into saltmarsh and the change of 16% of salt marsh into reedbed.

**Table 1. Matrix of land use change in Playa de la Hita wetland 1984-2001**

		2001						
1984	Land use/ land cover (ha)	Salt steppe	Salt Marsh	Reebed	Crops	Infra- Structures	Water Bodies	Total 1984 (ha)
	Salt steppe	0.88	1.63	0.00	0.06	3.00	0.00	5.57
	Salt marsh	0.13	13.94	3.50	0.06	3.88	0.00	21.51
	Reedbed	0.00	0.13	2.31	0.00	0.00	0.25	2.69
	Crops	0.13	1.13	0.06	0.00	2.19	0.00	3.51
	Infrastructures	0.06	0.00	0.13	0.00	4.75	0.00	4.94
	Water bodies	0.00	0.00	1.19	0.00	0.00	8.63	9.82
	Total 2001 (ha)	1.20	16.83	7.19	0.12	13.82	8.88	

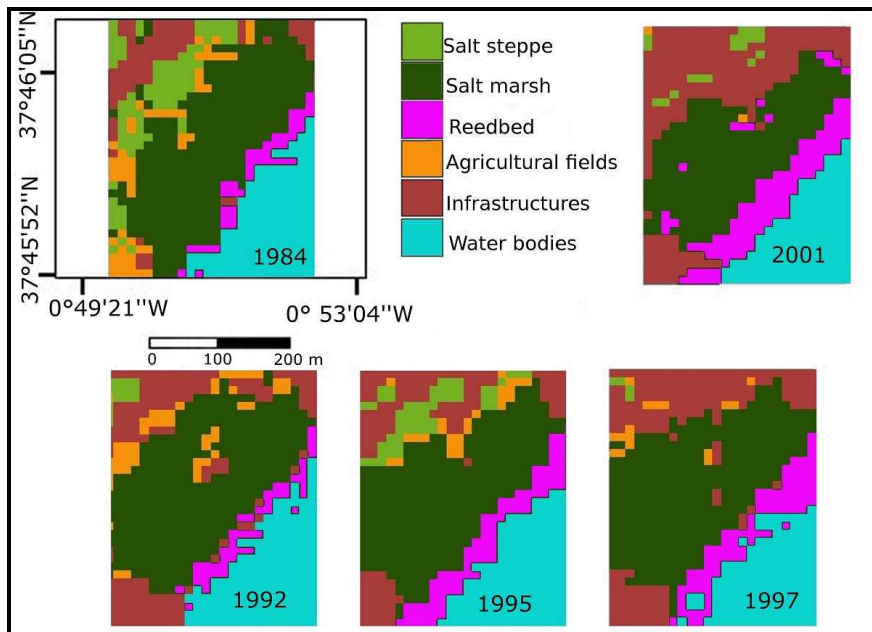


Figure 2. Land cover maps of Playa de la Hita wetland from 1984 to 2001 obtained by supervised classification of LANDSAT TM+ images.

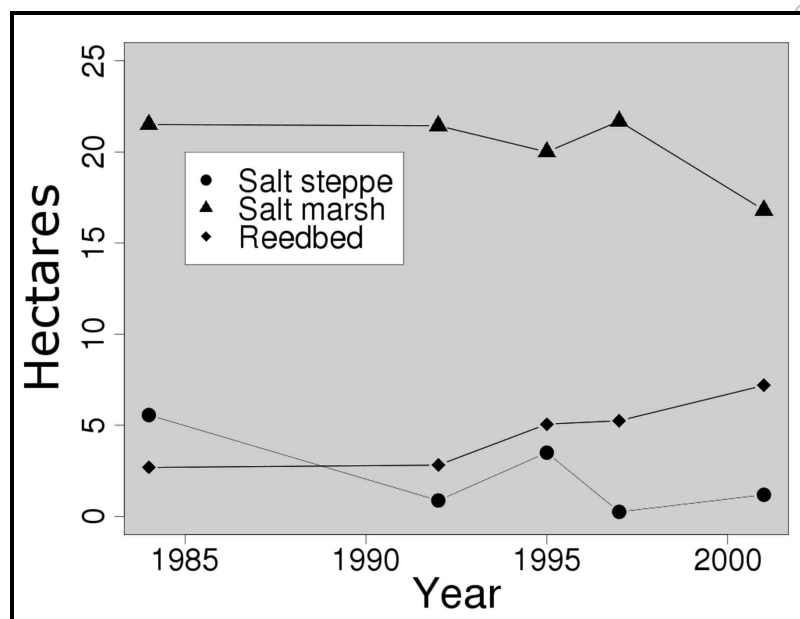


Figure 3. Area of salt steppe, salt marsh and reedbed in Playa de la Hita wetland from 1984 to 2001.

Taking into account all changes, salt steppe in 2001 had lost 78% of its area in 1984; saltmarsh had also lost area, although in a lesser degree (22%), whereas reedbed had increased by 167%. These trends of change are specially marked since 1995 (figure 3).

In overall, all these changes constitutes a loss of value from the Habitats Directive point of view, since it represents a significant loss of a Priority Interest habitat (salt steppe) and a loss in a Community Interest habitat (salt marsh) to the benefit of reedbed, not included in the Directive.

### *Saladar de Lo Poyo Wetland*

Figure 4 shows the land cover maps of Lo Poyo in 1984 and 2001.

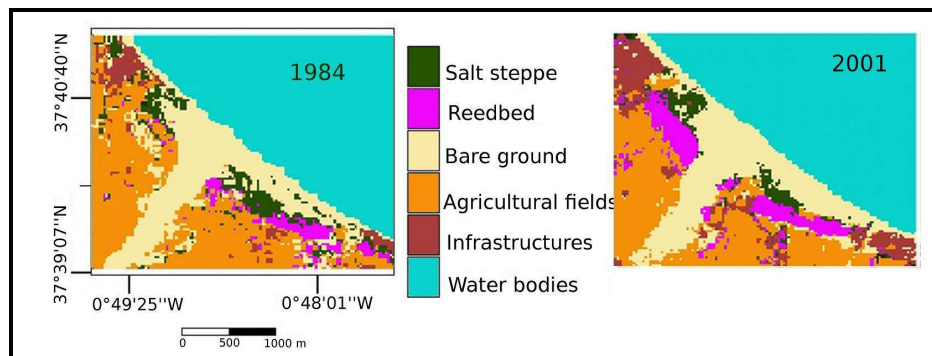


Figure 4. Land cover maps of Saladar de Lo Poyo wetland in 1984 and 2001 obtained by supervised classification of LANDSAT TM+ images.

From 1984 to 2001 there is a slight increase in the total area of the Saladar de Lo Poyo wetland due to the increase in reedbed. The two habitats present in Saladar de lo Poyo wetland (salt marsh and reedbed) shift in relative dominance between 1984 and 2001 (table 2): salt marsh occupies in 1984 a 63 % of total habitats area, whereas in 2001 66% of total habitats area is occupied by reedbed. These trends of change, similarly to the case of Playa de la Hita, also points to a loss of value from the Habitats Directive point of view.

**Table 2. Matrix of land use change in Saladar de Lo Poyo wetland 1984-2001**

		2001						
1984	Land use/ Land cover	Salt marsh	Reedbe d	Bare ground	Crops	Infrastructur es	Water bodies	Total 1984 (ha)
	Salt marsh	14.19	7.06	11.81	6.69	6.19	0.00	45.94
	Reedbed	1.94	11.56	2.25	6.88	3.88	0.00	26.50
	Bare ground	7.06	7.31	130.56	20.56	5.19	3.63	174.31
	Crops	5.00	28.13	12.06	139.25	26.00	0.00	210.44
	Infrastructures	0.38	0.69	5.38	6.44	15.75	0.00	28.63
	Water bodies	0.00	0.00	3.38	0.00	0.00	237.50	376.88
	Total2001 (ha)	28.56	54.75	165.44	179.81	57.00	377.13	

### *Marina del Carmolí Wetland*

Figure 5 shows the land cover maps of Marina del Carmolí between 1984 and 2001. Total wetland area increased by 12% between 1984 and 2001 and the habitat composition suffered a dramatic change. Marina del Carmolí was basically a salt steppe of 237 ha in 1981 (table 3) whereas in 2001 this habitat had lost half of its initial area and salt marsh and reedbed,

practically absent in 1984, occupy a significant extension (82 and 64 ha, respectively). Again, these trends of change implies a loss of value from the Habitats Directive point of view.

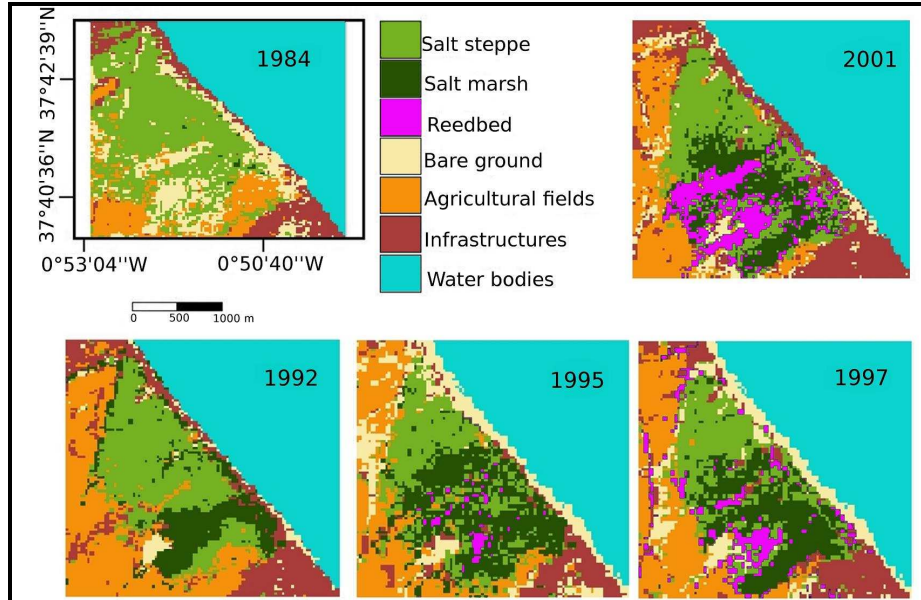


Figure 5. Land cover maps in Marina del Carmolí wetland from 1984 to 2001 obtained by supervised classification of LANDSAT TM+ images.

The pattern of change of the Marina del Carmolí habitats along time (figure 6) suggest two different periods. Between 1984 and 1995 there is a significant loss of salt steppe to the benefit of salt marsh. Between 1995 and 2001 the area of salt steppe is stabilised whereas there is a reduction of salt marsh associated to a steadily increase in reedbed. Therefore, the conversion of part of the initial salt steppe into reedbed, shown in table 1, is mediated by an intermediate stage as salt marsh.

**Table 3. Matrix of land use change in Marina del Carmolí wetland 1984-2001**

		2001							
1984	Land use/ Land cover	Salt steppe	Salt marsh	Reedbed	Bare ground	Crops	Infrastruc tures	Water bodies	Total 1984
	Salt steppe	91.75	49.79	23.13	10.19	41.94	20.44	0.00	237.22
	Salt marsh	0.00	2.03	0.00	0.00	0.00	0.00	0.00	2.03
	Reedbed	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
	Bare ground	18.00	12.25	29.63	10.75	11.06	20.25	0.19	102.13
	Crops	9.63	17.25	10.44	5.25	38.31	8.50	0.00	89.37
	Infrastructures	2.81	0.25	0.94	9.06	6.06	52.31	0.75	72.19
	Water bodies	0.00	0.00	0.00	1.31	0.00	5.44	295.44	302.19
	Total 2001 (ha)	122.19	81.56	64.13	36.56	97.37	106.94	296.38	

The relative changes between salt steppe, salt marsh and reedbed might be explained by the interaction between the soil moisture and conductivity gradients, as show data taken on 2003 and 2004 in the Rambla del Miedo area, one of the ephemeral channels entering into



Marina del Carmolí (figure 7). The regression model ( $R^2_{adj.} = 0.78$ ;  $p < 0.001$ ) shows that conductivity presents a quadratic response to soil moisture: at low values of soil moisture, conductivity increases with higher water content until a certain threshold, around 30% of soil moisture, above which conductivity decreases as soil moisture increases.

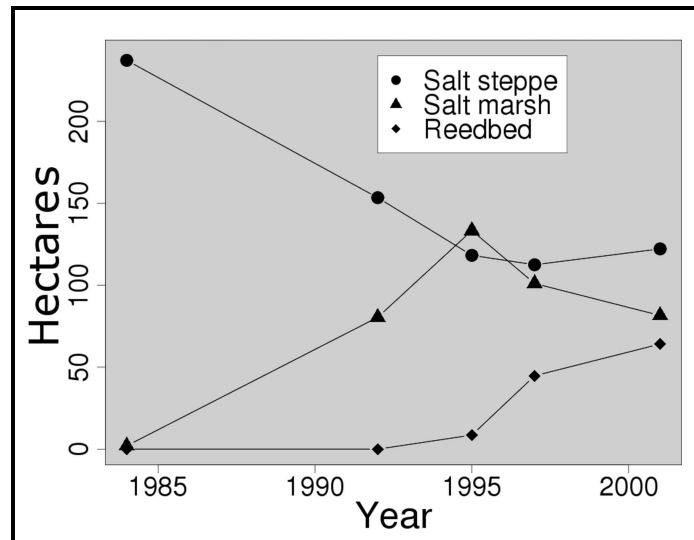


Figure 6. Area of salt steppe, salt marsh and reedbed in Marina del Carmolí wetland from 1984 to 2001.

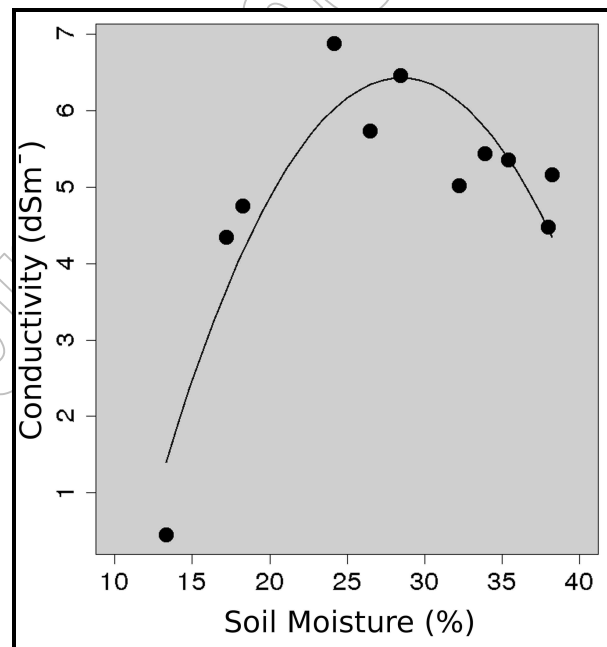


Figure 7. Data and regression model of soil moisture respect to conductivity in the Rambla del Miedo area, in Marina del Carmolí.

Figure 8 shows the spatial expression of this complex gradient along a transect crossing Marina del Carmolí in direction NW-SE, with the first and last samples located in the boundaries of the wetland. Samples with lower values of soil moisture, located far from the Miedo and Miranda ephemeral channels, present positive correlation between soil moisture and conductivity, whereas the samples located close to such watercourses, with higher soil moisture, presents a negative correlation between both variables.

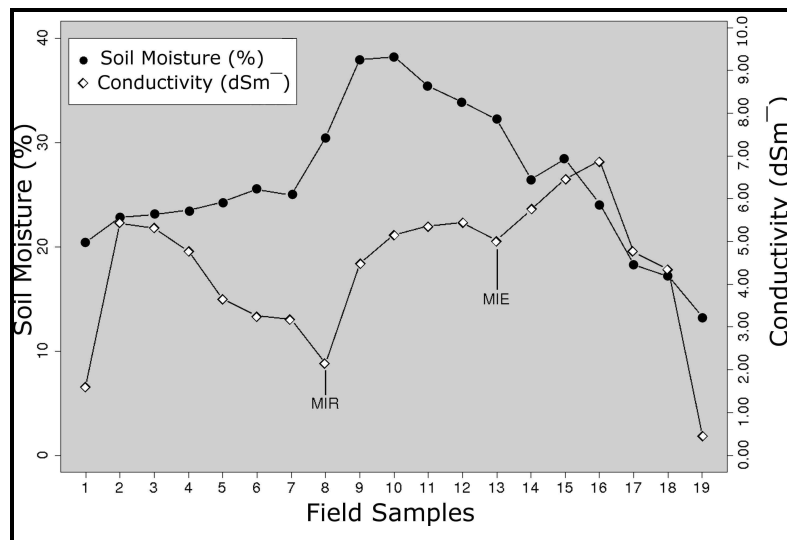


Figure 8. Soil moisture and conductivity along a transect crossing the Miedo and Miranda ephemeral channels, in Marina del Carmolí. Location of Miedo (MIE) and Miranda (MIR) ephemeral channels are indicated.

In synthesis, the initial increase in the water flows affecting Marina del Carmolí may have favoured higher soil moisture and therefore higher conductivity, what might have caused the increase in salt marsh at the expense of salt steppe. At a later stage beginning around 1995, greater water inputs would have caused a decrease in conductivity and allowed the extension of reedbed, as shown in figure 6. To this process may have also contributed, although with secondary importance, the inputs of urban wastewater into the ephemeral channels reaching Marina del Carmolí and some channelling works in the watercourses, which favour the spread of reedbed (Carreño *et al.*, 2008).

Although conductivity and soil moisture data before 2003 are not available, we have used the 2003-2004 data and the area occupied by each habitat between 1984 and 2001, obtained from the correspondent land cover maps, to estimate the change along time of these variables in Marina del Carmolí. Figure 2 shows the characterisation of the habitats in the wetlands (Playa de la Hita, Saladar de Lo Poyo and Marina del Carmolí) in terms of soil moisture and conductivity, respectively. The salt marsh presents the highest values of conductivity although the ranges of the three habitats partially overlap. On the contrary, salt steppe, salt marsh and reedbed are clearly differentiated along a gradient of increasing soil moisture.

Figure 10 presents a different illustration of the explained process. It has been calculated the weighted average of cuantile 25 of conductivity and soil moisture in Marina del Carmolí for each year, using the value of this parameter in the salt steppe, salt marsh and reedbed in

2003-2004 (figure 9) and the relative proportion of each habitat in Marina del Carmolí from 1984 to 2001, derived from the correspondent land cover maps. Results show a pattern coherent with the complex gradient between soil moisture and conductivity shown in figures 7 and 8. As shown in figure 10, the estimated quantile 25 of soil moisture and conductivity in Marina del Carmolí increases from 1984 to 1995, after which the further increase in soil moisture is accompanied by a decrease in conductivity, period in which the expansion of reedbed takes place.

In synthesis, the increase in agricultural drainages has caused important changes in the wetlands associated to the Mar Menor lagoon, both in total area and internal composition. There is an increase in 35 ha in natural vegetation (salt steppe, salt marsh and reedbeds), which occurred in Marina del Carmolí and Saladar de Lo Poyo, whereas in Playa de la Hita there was a slight reduction in the area of natural vegetation due to the construction of infrastructures.

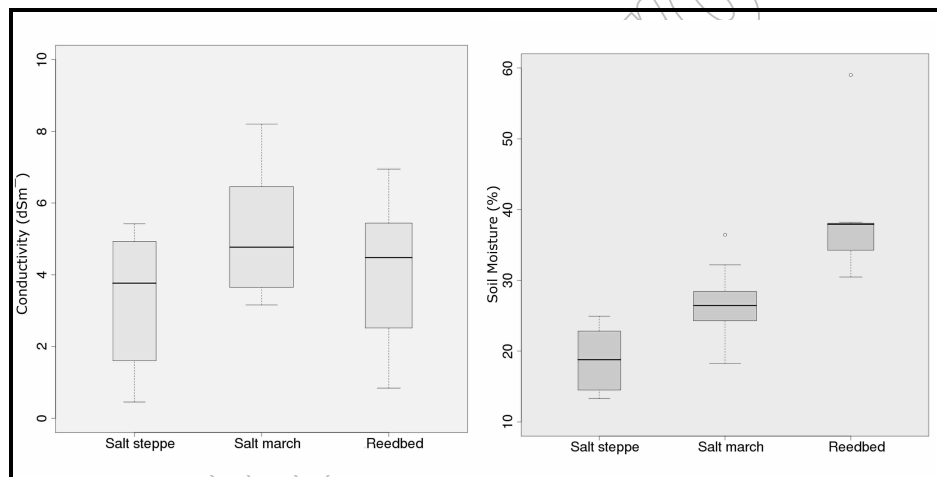


Figure 9. Ranges of conductivity (left) and soil moisture (right) of habitats in the Mar Menor wetlands. Boxes represent the quantiles 25, 50 (median) and 75.

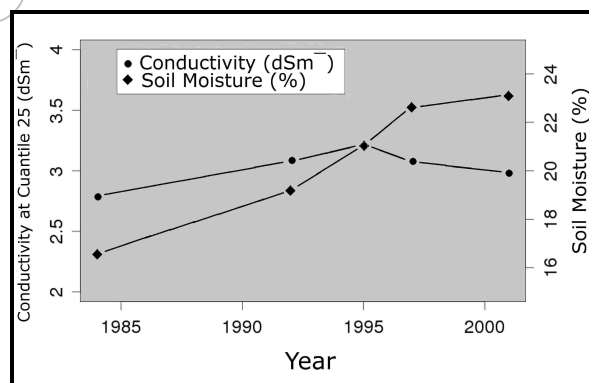


Figure 10. Estimated change in conductivity and soil moisture in Marina del Carmolí wetland between 1984 and 2003 (see text for details).

Regarding the internal composition, the habitats with higher requirements of soil moisture (salt marsh and reedbeds) have doubled and quadrupled their area, respectively, while the salt steppe has decreased to a half. The expansion of reedbed associated with shallower soil water conditions, has also been reported in other studies in the Marina del Carmolí wetland (Alvarez-Rogel et al., 2007b).

The net loss of salt steppe is very relevant, since it is the habitat with the highest interest from the Habitats Directive point of view (Priority Interest). Moreover, salt steppe is a rare habitat with a total area in Spain of only 12.976 ha, of which no more than 37% presents a good conservation state (Esteve and Calvo, 2000). Therefore, any reduction in the area of this habitat constitutes a significant loss, especially taking into account that in Murcia province, where the Mar Menor is located, the conservation state of this priority habitat is well over the average in Spain, with 83% of salt steppes in good conservation status (Esteve and Calvo, 2000).

It has been calculated an index to quantify the changes in overall value of wetlands from the Habitats Directive point of view along the period 1984-2001, as weighted average taking into account the area of each habitat and assigning the values 2, 1 and 0 to the Priority habitat (salt steppe), Community interest habitat (salt marsh) and rest, respectively. Figure 11 shows the sustained decrease (reduction in the index around 38%) in the overall value of the wetlands from the Habitats Directive point of view.

Land use changes at watershed scale are the primary factor explaining the described changes in the habitats of Mar Menor wetlands. The expansion of irrigated lands (Martínez et al., 2005; Velasco et al., 2006; Carreño et al., 2008) has doubled the irrigation water volume, causing the consequent increase in the drainage flows, part of which reach the Mar Menor wetlands. This has been confirmed by the rise in water tables in the aquifers of the watershed (ITGE, 1994; García Lázaro, 1995).

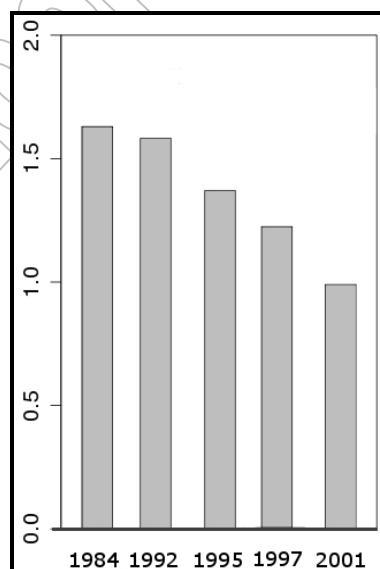


Figure 11. Value of the Mar Menor wetlands from the point of view of the Habitats Directive from 1984 to 2001, using the priority index (see text for details).

These changes have increased the levels of groundwater, flooding periods and soil water content in the wetlands, as reported in other field studies (Alvarez-Rogel et al., 2007b).

The role of the agricultural drainage coming from the watershed on the changes in the wetlands is supported by the close relationship between the sum of salt marsh and reedbeds in the wetlands and the area of irrigated lands in the watershed (Martinez and Esteve, 2002). Both variables show a similar pattern (Figure 12), characterised by sigmoid growth, which begins to slow down in 1991 in the case of irrigated lands and five years later in the case of salt marsh plus reedbeds. The regression analysis between both variables (figure 13) reaches a high significance ( $R^2_{adj.} = 0.917$ ;  $p = 0.01$ ).

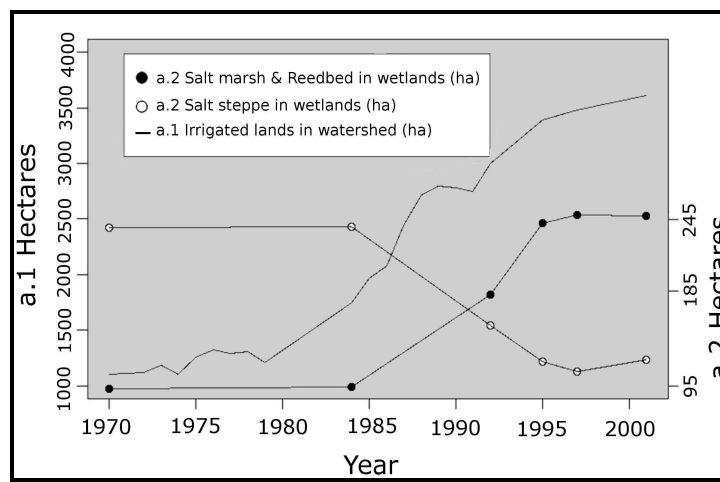


Figure 12. Temporal pattern of irrigated lands in Mar Menor watershed and area of habitats in the Mar Menor wetlands.

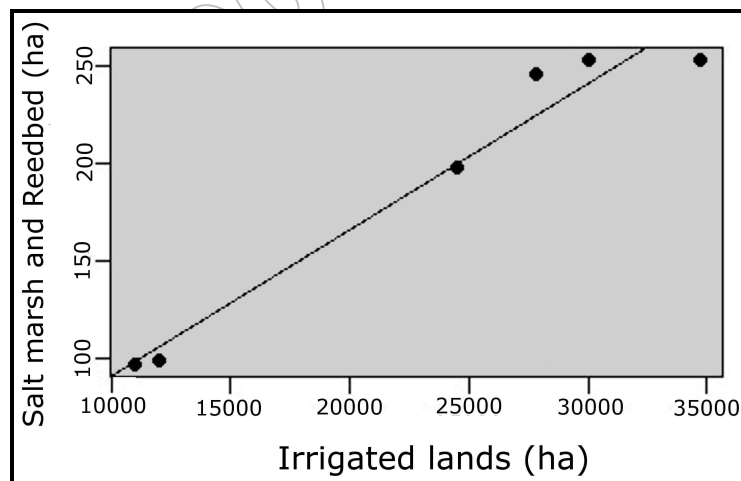


Figure 13. Regression model of the area of salt marsh and reedbed in the Mar Menor wetlands respect to the area of irrigated lands in the watershed.

When considering a five year time lag the relationship substantially improves ( $R^2_{adj.} = 0.945$ ;  $p < 0.001$ ), what might be interpreted as the time lag required by the surface and subsurface drainage flows to reach the wetland and by the habitats to respond to the increase in such water inputs.

It should be noted that the increase in the total wetland area does not appear to be a good indicator of the hydrological changes that have taken place in the watershed, since local conditions, in particular the spread of infrastructures like in Playa de la Hita, may limit the overall expansion of the wetland. On the contrary, the changes in the internal composition of each wetland towards more hygrophilous vegetation seems to be a good indicator of the hydrological changes at watershed scale, since there are no physical limitations to its expansion inside the wetland.

In synthesis, the long-term land use changes in the watershed, in particular the spread of irrigated lands and its effects on the hydrological dynamics, have important effects on the Mar Menor wetlands, of which the area occupied by salt marsh and reedbeds constitutes a good indicator. These changes implies a loss of value from the Habitats Directive point of view. In the next section we try to analyse whether faunistic assemblages, in particular wandering beetles, steppe passerines and waterbirds, are also affected by the changes in the water regime of the watershed.

## CHANGES IN WANDERING BEETLE ASSEMBLAGES

### Introduction

In this section we analyse if the described changes in the Mar Menor watershed have also modified the beetles community of the wetlands, in particular in Marina del Carmolí.

Carabidae and Tenebrionidae beetles are well known families that have frequently been used in ecological studies (Dajoz. 2002). Beetles of the former family are known to show measurable responses to environmental disturbance and degradation (Desender et al. 1994; Brandmayr et al.; 2000; Rainio and Niemelä. 2003), thus complying with the definition of an ecological indicator as described by Niemi and McDonald (2004). Beetles are therefore widely used in studies on habitat conservation (Eyre and Luff. 2002). Tillage, pesticide treatments, harvesting and other agricultural practices can cause disturbances (Cole et al., 2002; Holland, 2002; Lövei and Sunderland, 1996; Serrano et al., 2005) and significant changes in beetle assemblages (Belaoussoff et al., 2003). However, little attention has been paid to changes in the assemblages of wetlands adjacent to cultivated areas as a result of fluctuations in water tables related to agricultural management practices. Fuellhaas (2000) demonstrated that a rise in the water table led to the recolonisation of wetlands by carabids with good dispersal capabilities. Similar results were reported by Främbs (1990) in North-European wetlands.

Tenebrionid beetles have also been used as environmental bioindicators (De los Santos, 1983), particularly in arid environments such as those commonly found in the Mediterranean Basin. The loss of these species in arid Mediterranean systems has been related to processes of environmental degradation (Cartagena and Galante. 2002). Some species with a wide ecological tolerance are able to colonise peripheral wetland habitats with varying degrees of

soil moisture (Bujalance et al., 1987; Giménez and Esteve, 1994). The combined study of beetles of both families may therefore be useful to assess changes caused by variations in the water table and soil salinity. To this end, De los Santos (1983) proposed an index based on the Carabidae/Tenebrionidae ratio to evaluate the effects of climatic factors on beetle assemblages in Mediterranean areas. In this work we have studied the Carabidae and Tenebrionidae families to analyse their response to changes in the water and salinity conditions of the wetlands.

### Study Area and Methodology

In year 2003 the wandering beetles assemblages in ten sites located in the set of Mar Menor wetlands and peripheral areas were sampled with pitfall traps. Four sites, located in the Marina del Carmolí, were also sampled in 1984 with the same methodology, what allows a long-term survey in this wetland and an assessment of the changes in the beetles community along this period.

Three sampling sites were located in the inner parts of the wetland (W1, W2 and W3), while another another site (P1) was located in peripheral areas close to the wetland (figure 14). This last site showed no marked vegetation changes during the study period (1984-2003) and so no significant changes in beetle assemblages were expected in this peripheral site.

The survey also included data of soil moisture and conductivity in 2003 in the four sampling sites. Table 4 summarises the plant communities, soil moisture and conductivity in the sampled years and sites.

In each year the sampling period lasted six months, from 1 April to 30 September since, according to Esteve (1987) and Giménez (1999), the period of maximum activity of most of these species in wetland areas corresponds to spring and summer.

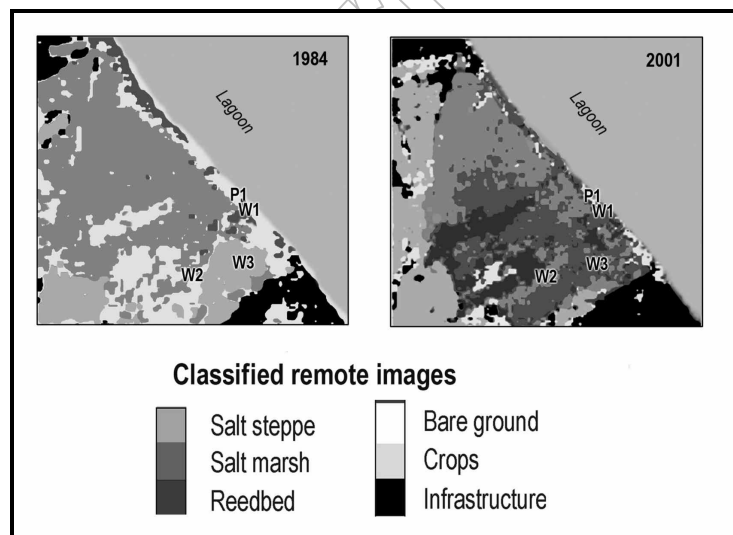


Figure 14. Location of the sampling sites in Marina del Carmolí wetland. W1, W2, W3: wetland sites; P1: peripheral site.

**Table 4. Details of the sampling sites of the Marina del Carmolí wetland (SE Spain). W= Sites deeply located in the wetland; P= peripheral site**

Site	1984	2003	Soil moisture 2003 (%)	Soil conductivity 2003 (dS/m)
W-1	salt marsh	salt marsh	29.78	7.28
W-2	salt steppe	reedbeds	38.22	5.16
W-3	cultivated	salt marsh	24.13	6.88
P-1	sands	sands	3.15	0.16

This has also been found to be the case in other beetle studies in arid systems (Yaacobi et al., 2007). Previous sampling campaigns in the area showed that any loss of information with regards to composition and abundance of Tenebrionidae and Carabidae in the Mar Menor wetlands is negligible. The traps consisted of plastic bottles of 1 litre capacity and 10 cm mouth diameter, filled with a saturated solution of NaCl. In each sampling site there were two rows of five traps each. Traps were emptied every 45 days (i.e., four times per year).

Beetles were identified to species level. The inventories and abundance data were used to calculate the species abundance per site and year. Population abundance is described through a capture-effort index consisting of the number of captures of each species per 10 active pitfalls during the six months sampling period. We calculated the logarithm of Carabidae/Tenebrionidae ratio and also the normalised Carabidae/Tenebrionidae ratio, which takes values between  $-1$  and  $+1$  (Pardo *et al.*, in press), depending on the relative predominance of tenebrionidae or carabidae, respectively.

Wandering beetles were classified as halobionts, halophiles and other preferences, according to their adaptation to and tolerance of salt and water in the soil (Serrano *et al.*, 2002). Briefly, halobionts are defined as species linked to salty soils that are rarely found in other environments. Halophiles are defined as those able to live in sites with a widely ranging salt content and thus must have efficient physiological mechanisms to maintain their water balance. The third group is made up of species with heterogeneous preferences (psammophiles, xerophiles, ripicoles) that are usually found at the periphery of saline habitats. This classification allows the response of communities to changes in the soil water content and salinity to be characterised.

## Results and Discussion

At all wetland sites there was an increase in the abundance of carabids and a decrease in tenebrionids (Figure 15), resulting in an increase in the normalised C/T ratio between 1984 and 2003 (Table 5). Peripheral site remained unchanged. The peak value corresponded to W2 in 2003, which also had the highest soil moisture.

This long-term change might be explained by the soil moisture changes caused by the modifications in the water regime at watershed scale, as discussed in the precedent section. Figure 3 shows the close relationship between soils moisture and logarithm of Carabidae/Tenebrionidae ratio using data from the ten sampling sites of the set of Mar Menor wetlands and peripheral areas.



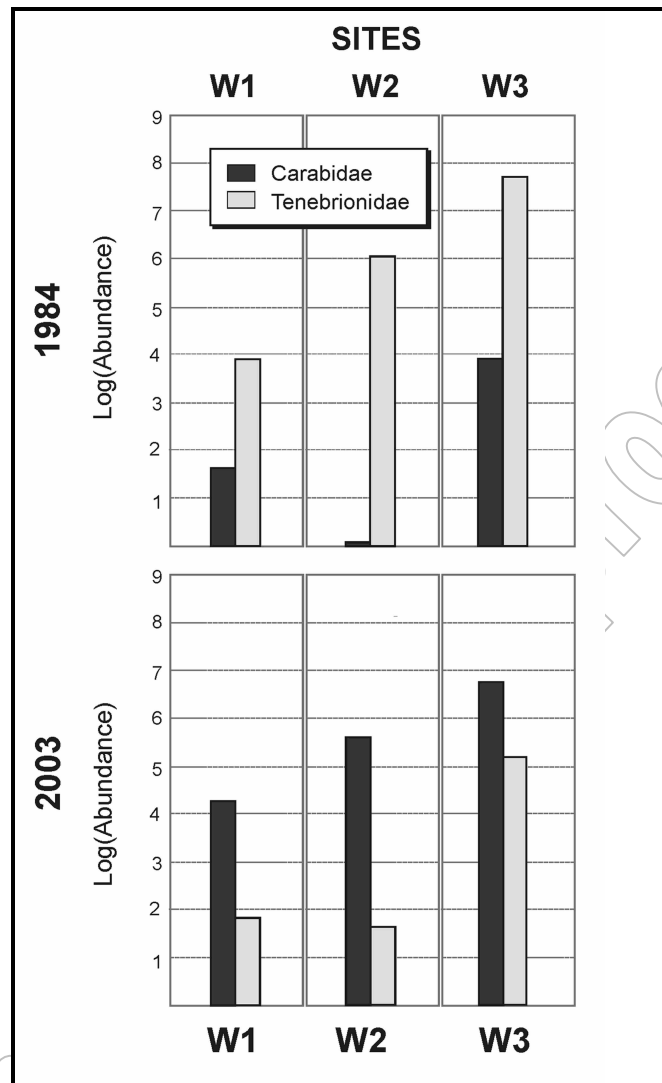


Figure 15. Abundance of carabids and tenebrionids in the wetland sites of Marina del Carmolí (W1, W2 and W3) in 1984 and 2003.

**Table 5. Normalised ratio of carabidae respect to tenebrionidae (C/T) in each site of Marina del Carmolí in 1984 and 2003. P1: peripheral site; W1. W2. W3: wetland sites**

Sampling site	1984	2003
P1	-0.9479	-0.9743
W1	-0.8148	0.801
W2	-0.9953	0.99446
W3	-0.9605	0.66145

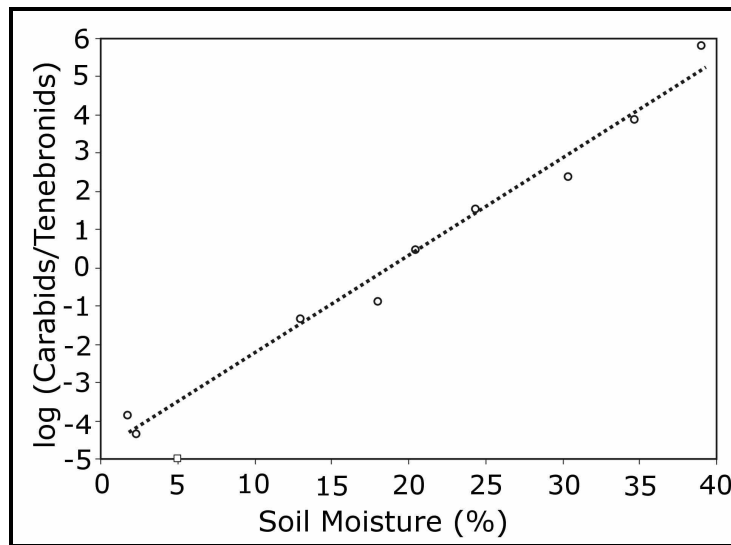


Figure 16. Logarithm of the Carabidae/Tenebrionidae ratio and soils moisture in the ten sampling sites of the set of Mar Menor wetlands and peripheral areas.

The effects of agricultural management practices on carabid assemblages through changes in the soil moisture content have been noted by Eyre et al. (1986). On the contrary, tenebrionid beetles were not favoured by an increase in soil moisture, as these beetles are better adapted to more arid environments and high soil moisture seems to be a limiting factor (Bujalance et al., 1987; De los Santos, 1983; De los Santos, 2002). Thus, a decrease in abundance of tenebrionid beetles could be a good descriptor of changes in the hydrological conditions of the wetland. Main tenebrionid species were *Tentyria laevis*, *Pimelia baetica* and *Zophosis punctata*.

Along with the change in the Carabidae/Tenebrionidae ratio there is a change in the composition of carabids. The assemblage of 1984 was dominated by xerophiles such as *Orthomus barbarus*, *Dixus sphaerocephalus* and *Ditomus tricuspoidatus*, whereas halobionts remained only in specific sites (W1). On the contrary, the assemblage of 2003 was dominated by halobionts and halophiles species, which seems to be favoured by the increase in the groundwater level, flooding period and soil moisture in the wetland sites in 2003, along with associated increased soil surface salinity (Figure 17).

The predominance of this type of species has been observed in littoral areas of saline lagoons (Rueda, 1990) and in small-scale gradients linked to the micro-relief of coastal dunes (Georges, 1999), which allow the presence of halophilous species in the intermediate levels susceptible to flooding. The halobiontic species were classified by Rueda (1990) as halophilic species adapted to floods and show medium to low body size, functional wings and high dispersal power. According to Den Boer (1987), these characteristics are typical of species inhabiting unstable habitats. In contrast, species that make up the 1984 assemblages were typical of more stable habitats.

Halobionts and halophiles dominated the assemblage in 2003 with species such as *Megacephala euphratica*, *Scarites procerus eurytus*, *Cylindera paludosa*, *Pogonus chalceus* and *Anisodactylus virens*. These last three species were absent in 1984. *Megacephala euphratica* and *Scarites procerus eurytus* (Figure 18) are the most singular carabid species

from a biogeographical point of view, as good examples of the Turano Mediterranean distribution pattern (Serrano *et al* 2002).

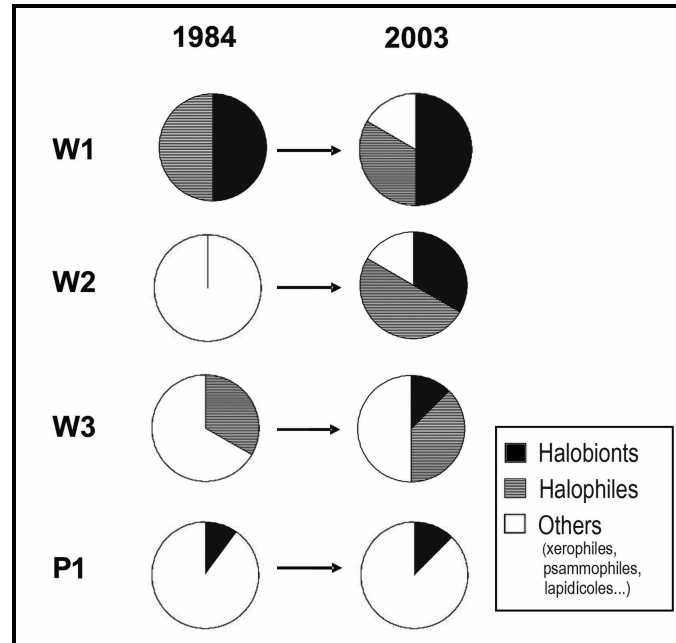


Figure 17. Proportion of halobionts, halophiles and groups with other environmental preferences in the sampling sites of Marina del Carmolí in 1984 and 2003.

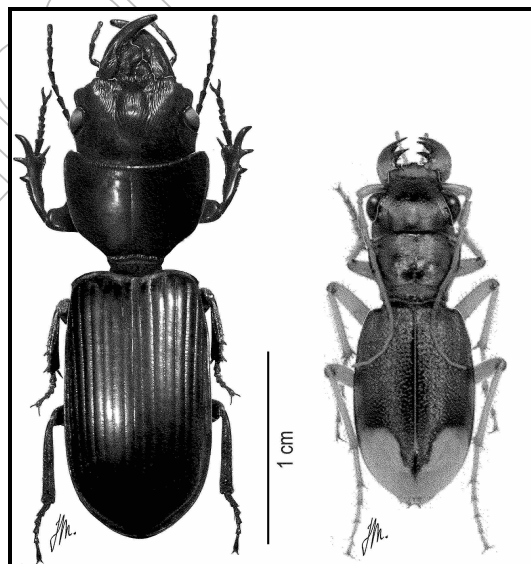


Figure 18. Left: *Scarites procerus eurytus*; Right: *Megacephala euphratica*.

These two species expand their distribution and /or multiply by twenty their abundances respect to 1984 (Table 6), as a response to the increase in soil moisture. Figure 19 shows the response model ( $R^2_{adj}=0.93$ ) of *Megacephala euphratica* to soil moisture and conductivity. The usual available habitat for *Megacephala* varies from more xerophilic salt marshes with 20% soil moisture and 18 dS/m conductivity to more humid salt marshes with 35% soil moisture and 7 dS/m conductivity.

In the peripheral site the situation was more stable, as xerophiles predominated from 1984 to 2003 and tenebrionid beetles, particularly core species such as *Tentyria laevis*, *Zophosis punctata* and *Gonocephalum rusticum* (Giménez, 1999), increased in abundance.

In synthesis, these results suggest that long-term trends in the groundwater table and soil moisture conditions in the wetland are well reflected in the changes in species composition at different taxonomic scales: proportion between families (Carabidae and Tenebrionidae) and the proportion between biological types within the dominant family under wetter conditions (halobionts and halophiles respect to groups with other environmental preferences).

**Table 6. Number of sampling sites with presence of *Megacephala euphratica* and *Scarites procerus eurytus* and average abundance of individuals per effort unit (10 pitfall traps) in Marina del Carmolí**

	1984		2003	
	NS	AA	NS	AV
<i>Megacephala euphratica</i>	2	0.75	2	14.80
<i>Scarites procerus eurytes</i>	1	0.25	3	6.50

NS: Number of sampling sites; AV: Average abundance of individuals per effort unit.

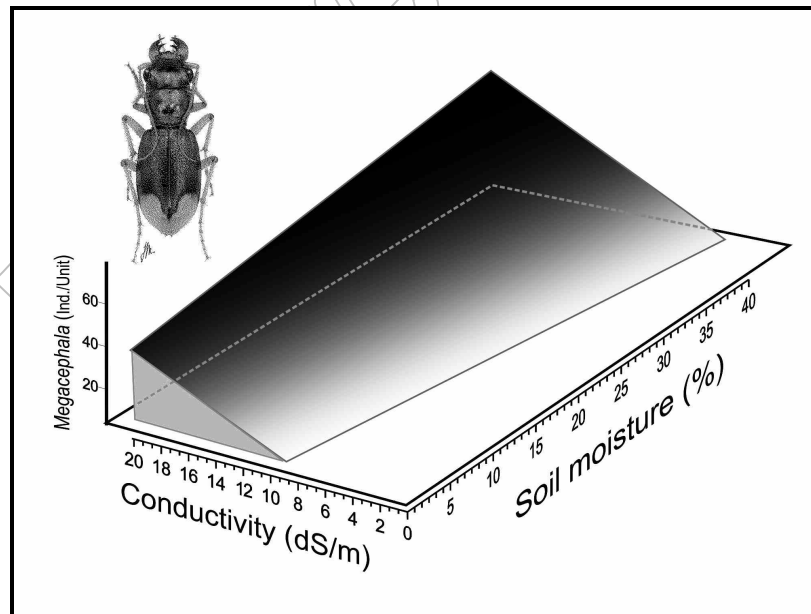


Figure 19. Response model of *Megacephala euphratica* to soil moisture and conductivity.

## CHANGES IN BIRD POPULATIONS AND COMMUNITIES

### Introduction

Surveys of bird populations and communities have been carried out in the Mar Menor lagoon and its associated wetlands since the early 70's of the past century. Previous pioneer studies (Guirao, 1859; Zamorano, 1932) 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 made in the framework of the International Waterbird Census (IWC) scheme, and 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 part of the chapter we focus on the two communities with the best series of data, when searching for biological indicators of landscape and environmental change in the wetland complex: steppe birds and waterbirds. As mentioned above, the latter have been surveyed through January census, more or less continuously since 1972 (Martínez et al., 2005; Robledano et al., 2008). For the former, we have been able to reconstruct a series of data extending from 1984 to 2008, through the compilation of several surveys undergone in the same wetland area (Marina del Carmolí) by the own authors and other researchers (Hernández, 1995; Torralva et al., 2003; Robledano et al., 2006).

### Study Area and Methodology

#### *Steppe Passeriform Birds*

The area surveyed is the Marina del Carmolí wetland, described elsewhere in this chapter. Census of passerines along line-transects in the salt steppe and salt marsh, were used to track changes in the terrestrial bird communities of the wetland. We used IKA values (birds/km) recorded in six years of transect sampling (spread along a period of 24 years): 1984, 1989, 1995, 1997, 2003 NS 2008. Dryland fields were sampled only in 1984. Seasonal averages (summer: April-September; winter: October-March) have been calculated when more than one transect census was available.

Line transects had a length of 0,5-1 km depending on the year, with the outer limit of the counting strip usually set at 40 m, on both sides of the transect. When a narrower sampling strip was used, the results have been corrected to refer to similar surface areas. Data for 2003 (Torralva et al. 2003), given in birds per hectare, have also been 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 track 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 four hours of daylight. The Marina del Carmolí (salt steppe and salt marsh) 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. 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 five years. We used several indices of abundance to relate changes in passerine populations and taxocenoses to habitat and landscape variables. 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 computed indices of conservation status, adapted from the procedure used by Pons et al. (2003) and ranked the 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). Separate analyses were performed on summer and winter community abundance matrices.

Abundance, richness, diversity and conservation status indices were related through simple linear regression with four predictor variables describing changes in the structure of vegetation and landscape characteristics, in turn related with the land-use changes in the watershed and the consequent changes in the water flows into the Marina del Carmolí wetland. These variables were the area of salt steppe, salt marsh, reedbed and crops in a 3 km<sup>2</sup> square window containing the wetland, as described earlier in this chapter. Mean IKA of passerine species and families and the community structural indices for 1984, 89, 95, 97 and 2003 were regressed on habitat variables for 1984, 92, 95, 97 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.

### **Waterbirds**

The study area for waterbirds is the whole Mar Menor water mass (figure 1), a 135 km<sup>2</sup> coastal lagoon located in the southeast of the Iberian Peninsula, with a mean depth of 4 m. We have used the results of the January waterbird censuses co-ordinated occasionally by the Regional Environmental Authority and most frequently by amateur environmental organisations (especially before 1995, and from 2000 onwards), the results of which have been compiled by Hernández and Robledano (1991), Martínez et al. (2005) and Hernández et al. (2006). The Mar Menor lagoon has the best coverage among the wetlands of Murcia Region, with 25 censuses between 1972 and 2005 (no census available for 1974, 1976, 1980-82, 1998, 2000-01 and 2003). Birds are censused through a standardised boat route that has remained approximately constant between years. Numbers counted of each species are pooled and reported together for the whole lagoon. We focus on the five most abundant waterbird species (Great Crested Grebe *Podiceps cristatus*, Black-necked Grebe *Podiceps nigricollis*, Great Cormorant *Phalacrocorax carbo*, Red Breasted Merganser *Mergus serrator* and

Common Coot *Fulica atra*), also characterised by a strong feeding association to the lagoon. With the exception of *Phalacrocorax carbo*, which can fly daily to carry out part of its feeding activity in nearby water surfaces (irrigation ponds and the sea), the individuals of these species spend all or a great proportion of their time budget inside the lagoon, and their feeding occurs typically in the water column and benthos. Individually or at the family level, they represent different foraging strategies, from specialist piscivores to generalist herbivores. Annual abundance (total number of each species in the January census) was converted into biomass using constant weight values obtained from the literature (Cramp, 1977). The percent contribution of each species, family or guild (for the Coot these were coincident) to total waterbird biomass was used to illustrate long-term changes in the trophic structure of the community. We interpreted these changes on the basis of the estimated nitrogen load reaching the lagoon the previous years, and two biotic variables: total adult jellyfish present in the lagoon, based on visual estimates and annual totals captured by boats committed to their removal, and the annual catch of the two main fish species exploited by the local fleet (*Atherina boyeri* and *Engraulis encrasicolus*), as a surrogate index of fish productivity.

## Results and Discussion

### *Response of the Steppe Bird Communities to Habitat Changes in the Carmolí Wetland*

Our compilation of terrestrial bird surveys shows dramatic changes in abundance and community structure, with important implications for the conservation value and function of this habitat for birds. In summer, total community abundance declined during the 24 year period of study ( $R^2_{\text{adj.}}=-0.6$ ;  $n=6$ ;  $p<0.05$ ), 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 the same or similar areas (Hernández, 1995). *Alaudidae* was the only family showing an overall significant decline with year ( $R^2_{\text{adj.}}=-0.81$ ;  $p < 0.01$ ), while there was an increase in the proportional abundance of *Fringillidae*, *Turdidae* and *Sylviidae* (Figure 20).

In winter, a similar pattern emerges 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 significantly increase with year ( $R^2_{\text{adj.}}=0.97$  and  $0.72$ ;  $p<0.001$  and  $0.05$ , respectively). The overall decline of *Alaudidae* is less pronounced than in summer yet also significant ( $R^2_{\text{adj.}}=-0.77$ ;  $p<0.05$ ), and there is a remarkable transient increase of *Fringillidae* in 1995-97.

Using data coming from so many different sources, however, 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 changes 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.

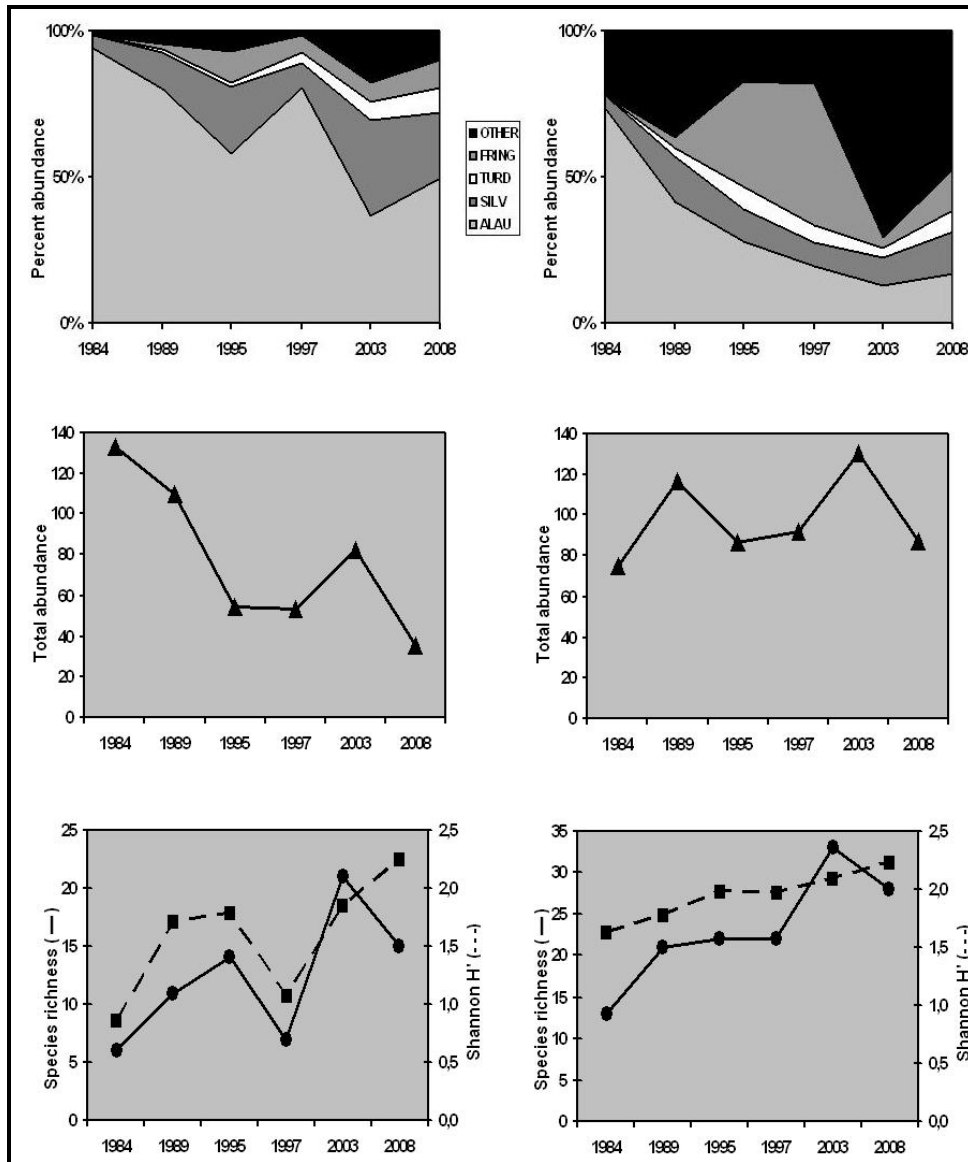


Figure 20. Changes in the relative abundance of the main families (expressed as % of total IKA), total community abundance and structural parameters of the passerine bird community (left, summer; right, winter).

The habitat variables most related with bird variation (Table 7) also suggest that summer and winter communities are affected by similar structural changes, driven by the pressures exerted on the steppe area by irrigation at the watershed scale. Many species did not respond individually to habitat variables, but representative species and families that can be considered habitat specialists did so. Figures 21 and 22 show some examples.



**Table 7. Response of some representative species and families to habitat variables**

		LINEAR REGRESSION (sign, adjusted R <sup>2</sup> and significance level)			
BIRD VARIABLES		STEP	SALT	REED	CULT
Summer community					
CALA	Melanocorypha calandra	0,99***	-0,76*	-0,72*	
COGU	Galerida cristata	0,72*	-0,69*		
CUTO	Sylvia conspicillata				0,76*
TACO	Saxicola torquata			0,94**	
TECO	Calandrella brachydactyla				0,80*
TRIG	Miliaria calandra				0,76*
VERD	Chloris chloris		0,61 (p=0,07)		
ALAU	Total Alaudidae	0,76*	-0,5 (p=0,11)		
TURD	Total Turdidae	-0,61 (p=0,08)		0,75*	
FRIN	Total Fringillidae	-0,49 (p=0,11)	0,85*		
TOTC	Total community	0,75*	-0,73*		
RDBO	Spanish Red Book Index				0,70*
Winter community					
ALRE	Lanius excubitor/meridionalis	-0,93**	0,79*		
BUIT	Cisticola juncidis				0,67 (p=0,05)
CALA	Melanocorypha calandra	0,63 (p=0,06)	-0,44 (p=0,13)		
COLT	Phoenicurus ochruros				0,68 (p=0,05)
ESPA	Emberiza schoeniclus			0,46 (p=0,11)	
ESTO	Sturnus unicolor			0,46 (p=0,12)	
GOCO	Passer domesticus				0,38 (p=0,15)
JILG	Carduelis carduelis			0,46 (p=0,11)	
MICO	Turdus merula			0,46 (p=0,12)	
MOCO	Phylloscopus collybita		0,46 (p=0,12)	0,46 (p=0,12)	
PETI	Erithacus rubecula			0,46 (p=0,12)	
PECH	Luscinia svecica			0,46 (p=0,12)	
PICO	Fringilla coelebs			0,46 (p=0,12)	
TACO	Saxicola torquata			0,70*	
TECO	Calandrella brachydactyla	0,59 (p=0,07)	-0,58 (p=0,07)		
ZOCO	Turdus philomelos			0,46 (p=0,12)	
ALAU	Total Alaudidae	0,66 (p=0,05)			0,48 (p=0,11)
TURD	Total Turdidae	-0,94**	0,88*		
HSHA	Shannon H'	-0,75**		0,52 (p=0,10)	
BDIR	Birds Directive Index	0,84*	-0,69*		
DESCRIPTOR VARIABLES		PEARSON CORRELATION COEFFICIENTS			
STEP	Area of salt steppe				
SALT	Area of salt marsh	-0,92*			
REED	Area of reedbed	NS	NS		
CULT	Area of crops	NS	NS	NS	

Correlations among variables are also shown. Marginally significant but biologically meaningful relationships are also included. Bird variables are mean IKAs of species or families except for structural or conservation indices. Significance levels:  $p < 0,05 = *$ ;  $p < 0,01 = **$ ;  $p < 0,001 = ***$

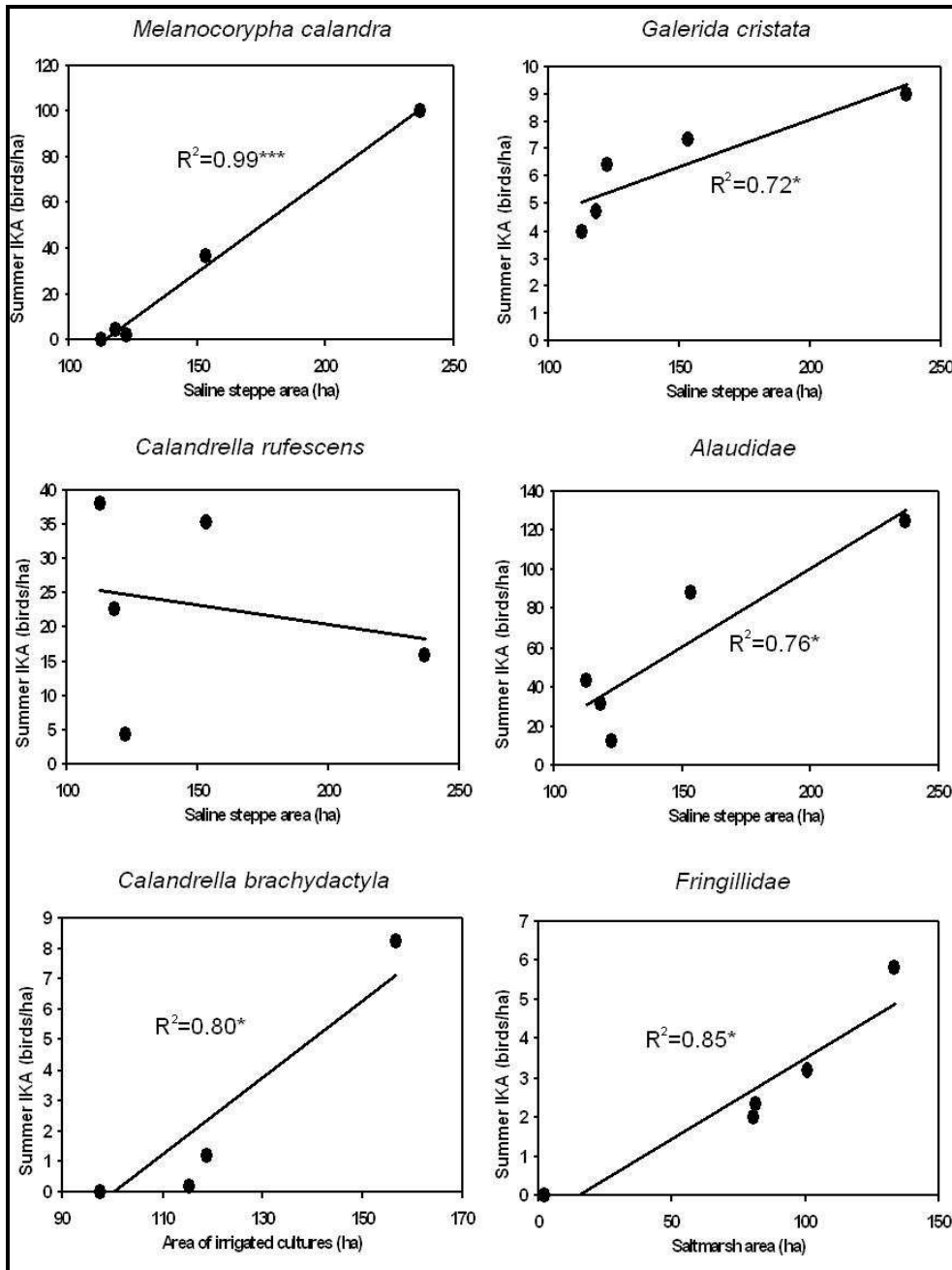


Figure 21. Response of some representative species and families of the summer community to habitat variables. *Melanocorypha calandra*, *Galerida cristata*, *Calandrella rufescens* and total *Alaudidae* are plotted against the area of salt steppe; *Calandrella brachydactyla* against irrigated lands and total *Fringillidae* against the area of salt marsh.

The indicator value of bird species, families and indices, regarding changes induced by agricultural influences, is illustrated by an early negative response of *Alaudidae*, particularly of Calandra Lark (*Melanocorypha calandra*) and Crested Lark (*Galerida cristata*) in summer,

and *M. calandra* and Short-toed Lark *Calandrella brachydactyla* in winter, to the loss of salt steppe in the habitat mosaic. Some species characterise the intermediate phases of change (by mid 1990's), when the loss of steppe habitat is stabilised and the development of salt marsh reaches its maximum, as described in the previous section). This is the case of *Fringillidae* in both seasons. *Sylviidae*, although favoured in general terms by the development of salt marsh, show different responses depending on the species, with Spectacled Warbler (*Sylvia conspicillata*) peaking in the early stages of change (1989), Dartford Warbler (*S. undata*) during the middle ones (1994-97), and Sardinian Warbler (*S. melanocephala*) towards the late ones (2003), probably illustrating species-specific habitat preferences. 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 salt marsh by reedbeds. In other wetlands, these species characterise the transitional (mixed) habitats located between the reed belt of lagoons and the salt marsh vegetation of immediate terrestrial areas (Peiró, 2006). The Lesser Short-toed Lark (*Calandrella rufescens*) shows also a pattern of intermediate increase in numbers (peaking between 1989-97). Its apparent indifference to any habitat variable can be due to the particular preferences of this species in terms of coverage and vertical structure of the vegetation. *C. rufescens* is common in coastal steppe areas with a good representation of salt marsh and low vegetation cover (Cramp, 1988; Tellería *et al.*, 1999; Torralva *et al.*, 2003). Hernández (1995) added to its preferences a weaker association with perennial grasses (e. g. *Lygeum spartum*, a dominant species in the salt steppe), and stronger with intermediate heights of these two types of vegetation (salt steppe and salt marsh).

The positive response of *Calandrella brachydactyla* (in summer) to the area of irrigated crops is somewhat unexpected since this species is usually favoured by dryland, unploughed agricultural habitats (De Juana and Suárez, 2004). In our study area it was more abundant in the peripheral dryland fields 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 the same season). An explanation for this is the particular nature of irrigated agriculture in this area. Due to the low quality of water, the consequent salinisation of soils, and the use of short crop cycles, the fields surrounding the wetland stay uncultivated for long periods, which favours the presence of many typical steppe species. In Murcia Region *C. brachydactyla* is a typical species of uncultivated agricultural fields Hernández (1995). In summer these fields seem to be playing a similar positive effect to that attributed by Brotons *et al.* (2005) to improved pastures around French steppes. The abundance of Corn Bunting (*Miliaria calandra*), a species of European concern (Birdlife International, 2004), is also positively related with the area of crops around the wetland. The recent breeding of Collared Pratincole (*Glareola pratincola*) in these areas, made possible through agreements between landowners and conservation organisations, is an example of the ecological potential of these areas, that should be also a priority conservation target. In winter *C. brachydactyla* seems to depend more on the salt steppe habitat.

It is not possible nor realistic, however, to try to explain most variation in all species or groups solely on the basis of local habitat changes. External factors, illustrated by population trends at higher geographical scales, can also explain the numerical changes of some bird species, particularly those showing cyclical phases of increase and decline in abundance. 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.

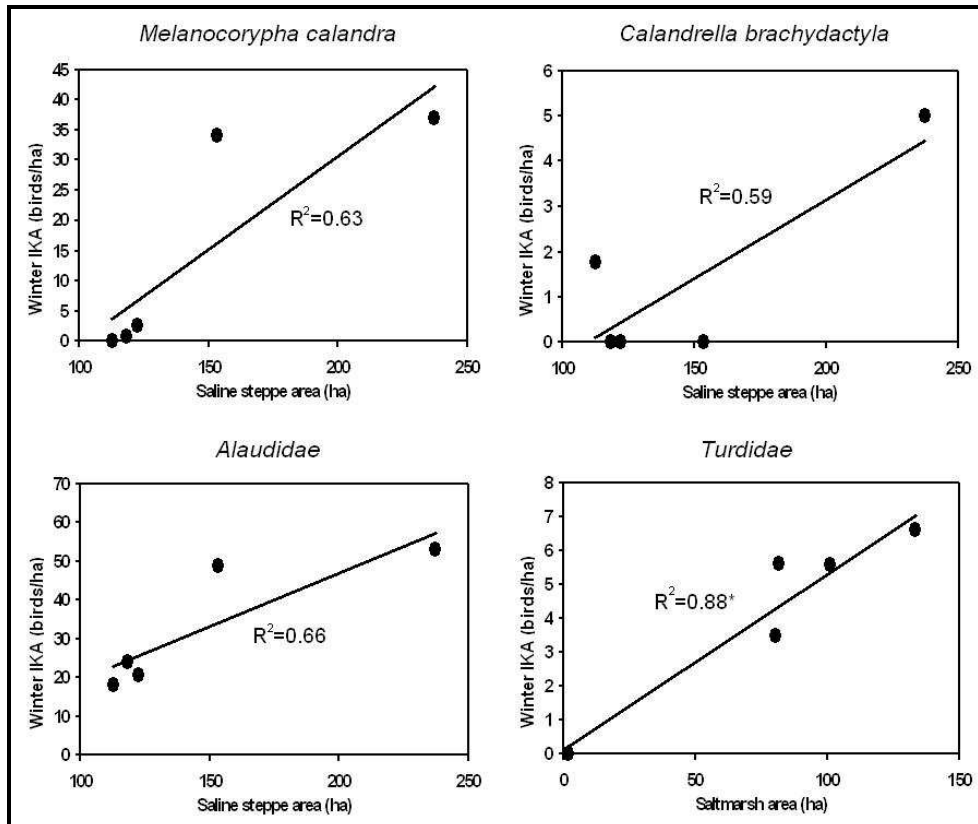


Figure 22. Response of some representative species of the winter community to habitat variables. Alaudidae species and total abundance are represented against the area of salt steppe and total Turdidae against the area of salt marsh.

Among the conservation indices, those based on SPEC and on the Spanish Red Data Book show an overall decrease in the summer community (although with a partial recovery in recent years), and fluctuant trends with even some gain at the end of the period. The Birds Directive index decreases both in summer and winter (Figure 23). The latter trend is the only significant one ( $R^2_{\text{adj.}}=-0.58$ ;  $p<0.05$ ). In both seasons, the decline in the Birds Directive index parallels that of the Annex I species present, namely *Melanocorypha calandra* and *Calandrella brachydactyla* in summer, and these same species plus *Sylvia undata* in winter. Apart from these, the general loss of conservation value in summer is due to the decline in species of open spaces and mosaic habitats like *Miliaria calandra* and *Lanius senator* (SPEC3). Among “winner” species adding conservation value we can remark *Emberiza schoeniclus* and *Carduelis cannabina* in winter. In overall, besides increasing species richness and diversity, the change towards less steppic conditions represents a small gain in conservation value. The species that benefit from this change are usually the less representative of the original conditions and often marginal to the region.

Although high values of species richness and diversity are often set as general targets in protected area management, the use of more specific conservation indices allow a best evaluation of performance against particular objectives. Our indices 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 greater concern should be put on the former. The Carmolí wetland has been declared as a Bird SPA (Special Protection Area) 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 (*Calandrella 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. Our estimates of summer population density for this species are 20 birds/10 hectares 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 y Pela, 1987; Suárez *et al.*, 2002; Samprieto and Pelayo, 2003).

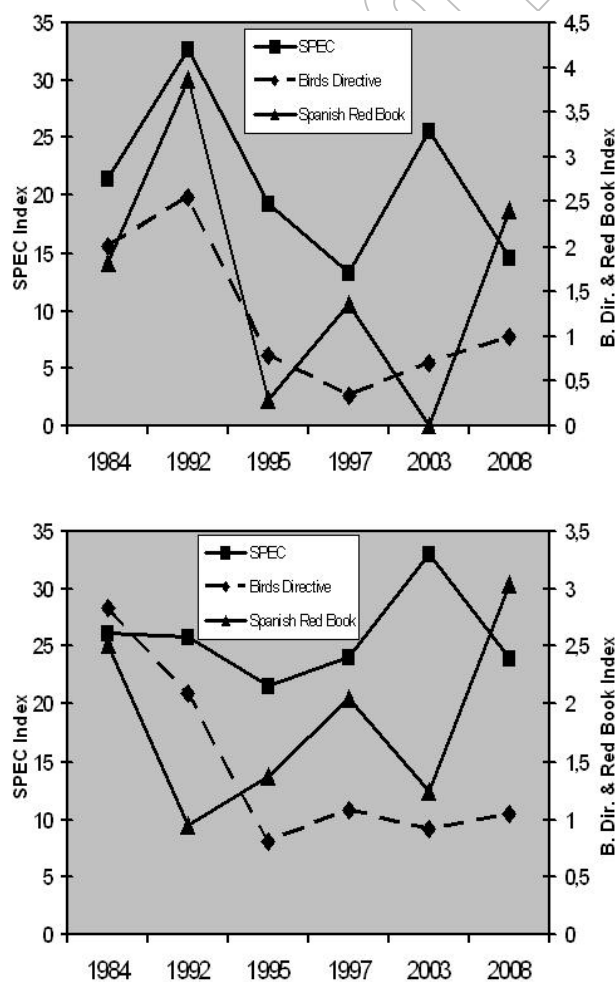


Figure 23. Changes in the indices used to assess the conservation value of the study area (upper graph, summer; lower graph, winter).

This transient positive effect can be attributed to the expansion of salt marsh vegetation which stabilises by 1997. In the long term, however, its abundance is lowered to values between 5.5-8.75 birds/10 ha, still high but nearly an order of magnitude lower than the peak values (on which the evaluation against the SPA criteria is based).

### **Dynamics of Waterbird Communities in the Mar Menor Lagoon**

Of the five species studied, all but the Red Breasted Merganser have increased in numbers along most of the period of study. The greatest fluctuations and highest counts correspond to the Black-necked Grebe. The trend of the Great Crested Grebe is parallel to that of its congeneric species, but less fluctuant. Cormorants seem to have grown more steadily. Coot display also a marked increase, but starting later than these three piscivorous species. Figure 24 summarises the general progression of waterbird biomass and the relative contribution of each species. Despite the gaps at the start and the end of the period, there are recognisable periods characterised by the dominance of particular species. The Red Breasted Merganser is the dominant piscivorous during most of the decade of 1970's. The Great Cormorant dominates most years since then, representing ca. 50% of the biomass except between 1988 and 1996, when other piscivores, including the two Grebes, dominate. Herbivores (Coot) join the community in 1992 and their contribution rises to more than 30% in 2005. The biomass of Red Breasted Merganser does not change markedly in the long term, although its relative contribution decreases gradually (from more than 90% to less than 15%). In terms of feeding guild numbers, the composition of the waterbird community has changed from a 100% of piscivores until 1991, to nearly 50% of herbivores in 2005. When expressed as biomass, the contribution of herbivores is slightly lower (under 40%).

The development of intensive agriculture and residential tourism during the last three decades have become the main pressures driving environmental change in the Mar Menor Lagoon. Total waterbird biomass has increased more than 4-fold during this period, which supports the increase in carrying capacity as a plausible explanation (Ysebaert, 2000; Van Eerden *et al.*, 2005; Roomen *et al.*, 2006). The positive response of bird biomass to estimated nutrient loads into the Mar Menor lagoon (Martinez *et al.*, 2005) is in accordance with this explanation, as does the lack of a general relationship with biogeographical population trends (F. Robledano *et al.*, in prep.). Local trends of bird numbers differ also markedly from those observed in other nearby sites or regions, in some species with opposite rather than divergent trends. This is the case of the Coot, which declined sharply as a breeder in the Valencia region during the period 1984-2004, and especially from 1991 onwards in the winter census, falling from more than 14,000 birds to less than 2,000 (Gómez *et al.*, 2006; compare this with the 10-fold increase in the Mar Menor between 1992 and 2005). The carrying capacity of the lagoon for this phytophagous species seems to have risen during the second half of this period, opposite to what happened in a number of apparently more suitable wetlands in the Valencia region. This is not surprising since, in their preferred sections within the Mar Menor lagoon, Coot attains higher densities than in brackish coastal wetlands of southeastern Spain with dense *Ruppia* meadows (Robledano *et al.*, 2008).

Nutrient load appears as the first determinant of waterbird biomass in three out of five species, with the dynamics of a fourth one (Great Cormorant) primarily governed by external factors (see e.g. Van Eerden and Gregersen, 1995; Van Bommel *et al.*, 2003) but also favoured by local eutrophication. On the other hand, Red Breasted Merganser seems to have been indifferent (tolerant) to nutrient loading during most of the study period, although

negatively affected in the long term (Martínez *et al.*, 2005). The trends of Great Cormorant and Red Breasted Merganser in the Mar Menor match best those of their Spanish populations (Martínez *et al.*, 2005), and consequently they could be considered as poorer indicators of lagoon condition.

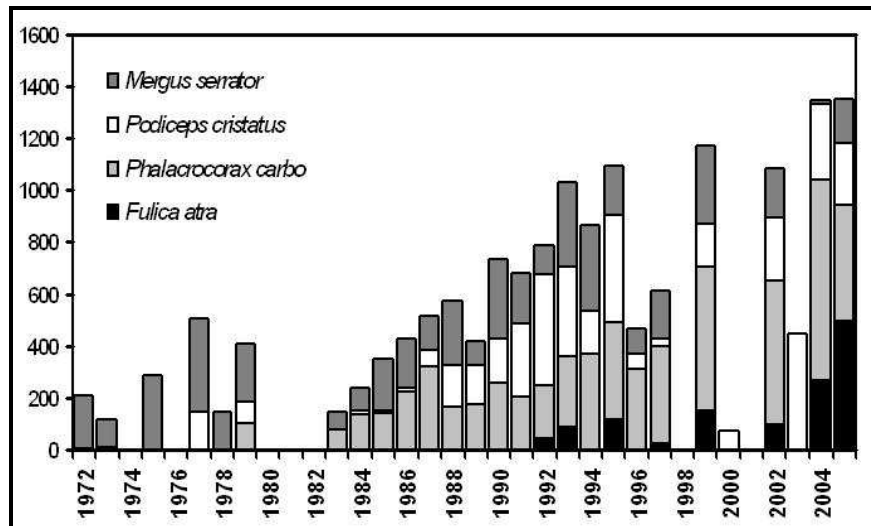


Figure 24. Contribution of the five species studied to the total biomass (kg) of the waterbird community of the Mar Menor Lagoon (only years with data for all the species altogether are shown).

The question remains about through which mechanisms eutrophication improves feeding conditions for waterbirds. Some authors (e. g. Nienhuis, 1992) argue against a direct relationship between eutrophication and higher secondary production. But at a general scale, the changes in waterbird numbers have been commonly related to the fertilisation of coastal waters, with increases and declines interpreted on the basis of processes of nutrient addition and removal (see references in Robledano *et al.*, 2008). Nutrient loads cause increased nutrient concentration that stimulates phytoplankton production and thereby increases chlorophyll concentration and light attenuation (Cloern, 2001). The effect of nutrient loads on fish biomass is also usually positive (Van Rijn and Van Eerden, 2003; Pérez-Ruzafa *et al.*, 2007).

Of course, factors other than agricultural nitrogen load can also explain changes in waterbird abundance. Phosphorus loads from urban sewage can have an important role in the eutrophication process. The general increase in productivity can also respond to climatic factors. Changes in the salinity of the lagoon can also account for some changes in waterbird populations. Long term variation in several other geomorphologic and hydrographic parameters, through its influence on fish community composition and abundance (PÉREZ-RUZAF A *et al.*, 2007), can also influence waterbird numbers. Weather (outside and in the wintering places) and disturbance are factors that can also affect waterbird populations (Davidson and Rothwell, 1993; Rönkä *et al.*, 2005; O'Connell *et al.*, 2007) and hence census results. It is also possible that variations in the winter pattern of occurrence of the species explain apparent interannual changes (Roomen *et al.*, 2006).

Aggregated variables (family, guild or total waterbird biomass) seem to respond more markedly than individual species' abundances to gross nutrient enrichment. An aggregated approach can also be applied to the temporal sequence of years, trying to identify phases of relative stability and sharp changes in the lagoon ecological status. Within the general increase in waterbird biomass that starts by mid 1980's, we can distinguish four main phases plus a short one of rapid change (Figure 25): Phase 1 and 2 correspond to a period of more or less homogeneous nutrient loading, the first part of which (1972-79) still shows high fish catches that dramatically drop at its end (probably due to overfishing); the second phase (1980-87) shows a first positive response of generalist piscivores, mainly Great Cormorant, which dominates the community with Red Breasted Merganser. It follows a period of gradually increasing nutrient loads extending approximately from 1988 to 1995 (Phase 3), during which the relative biomass of *Podicipedidae* (vs that of Cormorant plus Merganser) increases markedly. This is followed by a short phase (1996-98) with low fish catches, higher nutrient loads and an incipient jellyfish development (Phase 4), coincident with a fall in *Podicipedidae* biomass. From 1999 onwards, there is some recovery of *Podicipedidae* but their relative contribution to total biomass is lowered due to the large increase of Cormorant and the also increasing share of herbivores (Coot). This last period (Phase 5) is characterised by further increases in nutrient loads, but also by a greater abundance of jellyfish that can exert some control on eutrophication. In fact there is an apparent recovery of fish catches that can be related to such control, which in turn could be responsible for the recovery of piscivores after 1998. Towards the end of the study period, the continued increase in nutrient loads and the decline of jellyfish numbers could illustrate a new shift, in this case towards conditions more favourable for herbivores. Eutrophication causes a marked deterioration of seagrass and macroalgal communities, a situation apparently not beneficial for herbivores (Noordhuis *et al.*, 2002), but it also can promote a shift towards increased abundance of opportunistic macroalgae (Krause-Jensen *et al.*, 2008), a source of food for generalist herbivores like Coot (Perrow *et al.*, 1997). The proliferation of such algae is a phenomenon already observed in some stretches of the lagoon shoreline.

Surprisingly, four out of five waterbird species increase in numbers along a period of fisheries decline, and only one of them is typically non-piscivorous (Coot). However, it is possible that the piscivorous waterbirds select different prey types or size classes than those commercially exploited (Liordos and Goutner, 2007), which in the long term seem negatively affected by eutrophication. But the decrease in fish yield can also be a consequence of overfishing, leading to a dominance of small fish in the community due to the selective extraction of larger ones (Van Rijn and Van Eerden, 2003). This larger biomass of small fish is known to create good feeding conditions for species like the Great Crested Grebe and the Great Cormorant (Gwiazda, 1997; Smit *et al.*, 1997; Van Rijn and Van Eerden, 2003). Other species can survive entirely on prey other than fish, like Black-necked Grebe, a typical invertebrate-feeder (Jehl, 2001).

Interaction among variables (F. Robledano *et al.*, in prep.) suggests that some biotic components might have effects on the relationship between waterbirds and trophic variables. Particularly, jellyfish seem to modify the response of waterbirds to nutrient enrichment. Jellyfish populations are the main top-down agent in the Mar Menor lagoon, controlling the effects of the eutrophication process through two mechanisms: the direct use of nutrients by endosymbiotic zooxanthella, and the direct predation on plankton (Pérez Ruzafa and Aragón, 2003; Rodríguez *et al.*, 2005).



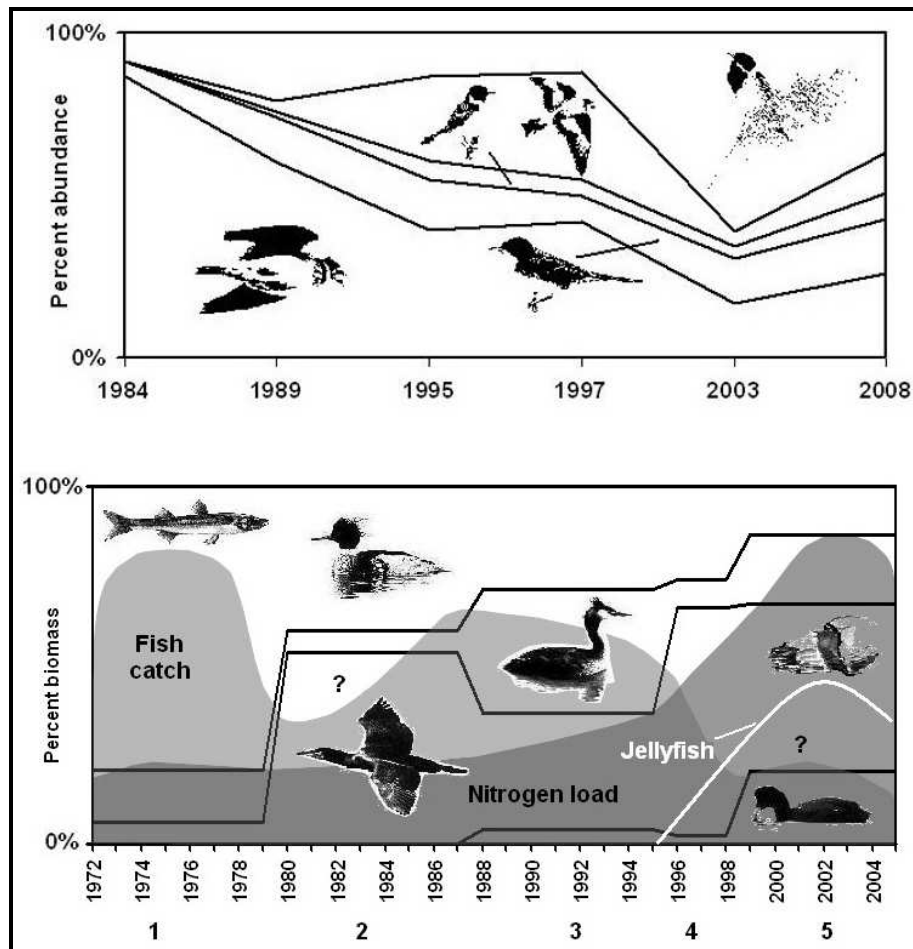


Figure 25. Simplified representation of the main phases identified by the response of terrestrial passerines and waterbirds to habitat and environmental variables. Species or families are represented by areas proportional to their contribution to total community abundance (passerines) or biomass (waterbirds), and environmental variables by curves or areas. Values are averages of summer and winter IKAs (passerines) and of January census for each period (waterbirds). Taxa represented are (from bottom to top): Waterbirds: Coot, Great Cormorant, Podicipedidae and Red Breasted Merganser; Passerines: Alaudidae, Sylviidae, Turdidae, Fringillidae and others.

Jellyfish play a role of sink for organic matter, by taking up and storing nutrients, and typically disrupt the trophic web at the level of secondary consumers. By feeding on organisms that serve as prey of large consumers like large crustaceans and fish larvae, they also reduce their feeding opportunities (Gili and Pagès, 2005). Increasing anthropogenic nutrient loads favours jellyfish, acting synergistically with overfishing to result in a degradation of the ecosystem, characterised by harmful algae and jellyfish blooms (Vasas *et al.*, 2007).

Summarising, long-term negative effects are only apparent for the Red Breasted Merganser, which seems to prefer low to moderately eutrophic conditions. The two *Podicipedidae* increase in moderately eutrophic conditions and keep their contribution to waterbird biomass along most of the study period, except during the short phase of higher

enrichment when jellyfish are still unable to alleviate eutrophication (Pérez Ruzafa *et al.*, 2002). However, we think that further nutrient addition, particularly in the absence of this top-down control (jellyfish numbers have been lowered to a minimum during the last removal campaigns), can lead the system to conditions less favourable for most piscivores. The feeding activity of great crested grebes in eutrophic waters has been shown to be limited by factors like underwater visibility (Van Eerden *et al.*, 2003). The Great Cormorant, also, has been proposed as an indicator of waters of intermediate turbidity, above (or below) which, it feeds less efficiently (Van Rijn and Van Eerden, 2003). Present favourable conditions for herbivores can also deteriorate in the long term if the system shifts to a phytoplankton-dominated state (Noordhuis *et al.*, 2002; García-Pintado *et al.*, 2007).

## CONCLUSION

We can conclude, answering the questions starting this chapter, that wetlands have effectively reacted in a strong manner to the land use changes in the watershed and associated alterations of the water and nutrient flows. Moreover, the changes can be tracked in different biological communities, in particular the vegetation assemblages, the carabid and tenebrionid communities, steppe passerines and waterbirds. It has been shown that the effects on steppe passerines are mediated by the changes in the vegetation assemblages, as shown by the regression models between the steppe passerines and the area occupied by each type of habitat along time. Some trends constitute a common pattern of change in all studied communities, especially the reduction of the elements more specifically linked to arid characteristics, like the area of salt steppe habitat, the abundance of tenebrionids and steppe passerine birds such as *Alaudidae*.

In each assemblage it has been identified some good indicators, not defined at the species level, of the observed long-term changes. Regarding the vegetation assemblages, the overall increase in the total wetland area seems to be a poor indicator of the increase in water input at watershed scale, whereas the increase in the hygrophilous vegetation (salt marsh plus reedbeds), observed overall and in each particular wetland, constitutes a good indicator of such water changes. A very significant relationship was found between the temporal trend of the area of irrigated lands in the watershed with a five years time lag and the area of salt marsh plus reedbeds in the wetlands. In the case of wandering beetles, long-term trends in the groundwater table and soil moisture conditions in the wetland are well reflected in the changes in species composition at different taxonomic scales: proportion between families (Carabidae and Tenebrionidae, as expressed by the Carabidae/Tenebrionidae ratio) and the proportion between biological types within the dominant family under wetter conditions (halobionts and halophiles respect to groups with other environmental preferences). In particular, the C/T ratio, which does not require identification of taxons to the species level, only to the family level, seems to be a very good indicator of the soil moisture to explain both spatial patterns and long-term changes. The value of terrestrial birds as indicators of land use changes is illustrated by their integrative, multi-stage response to the habitat changes. The main phases, defined by the relative proportions of habitats, are associated with numerical changes in species, families and assemblages. In our case the decrease in abundance of *Alaudidae* in the wetland seems to be a good indicator of the land use changes in the

watershed. Specific *Alaudidae* species seems to explain in more detail the nature of such changes. Waterbird lagoonal species, although having lower conservation scores (none of the species studied qualify for SPA designation), emerge as an equally valuable tool in the surveillance of agricultural (and urban) nutrient impacts on the lagoon. The warning role of waterbirds about changes in nutrient status in the eutrophication process of the lagoon (suggested as early as mid 1980's), could also have been played by steppe passerines with regard to habitat changes in the peripheral wetlands (evident by mid 1990's, 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).

Regarding the biodiversity and conservation value, different trends have been identified. In the case of the vegetation communities, the habitats favoured by the increase in irrigated lands and the water changes in the watershed are those less valuable from the Habitats Directive perspective. The habitat showing the highest increase is reedbeds, which is not included in the Directive, followed by the salt marsh, designated as Community Interest, at the expense of a substantial reduction in salt steppe, of Priority Interest due to its singularity and level of threat. This is relevant since it is a rare habitat in Spain and the conservation status of salt steppe in Murcia is well over the average in Spain. In the case of wandering beetles, the changes in the water regime have benefited the most singular carabid species from a biogeographical point of view, *Megacephala euphratica* and *Scarites procerus eurytus*, good examples of the Turano Mediterranean distribution pattern. In the case of steppe passerines, 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. However, the Carmolí wetland has been declared as a Bird SPA (Special Protection Area) under the Birds Directive, hence, the loss in conservation value of Marina del Carmolí according to the Bird Directive based index implies that the protection function for this natural site has not been achieved.

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 salt marsh and reedbed communities. The Mar Menor wetlands, in particular Marina del Carmolí, exemplifies the deterioration of a steppe area through hydrological changes in the watershed, without direct habitat reclamation. The need to manage agricultural impacts at the watershed scale emerges here as a critical biodiversity issue. Ecological changes induced by excess drainage water and nutrient loading into the Mar Menor lagoon and its peripheral wetlands have modified its biota, causing a loss of ecological integrity and conservation value. The observed trends, which will probably continue in the future unless action is taken in the whole area of influence, question the effectiveness of the protection measures given to the area and highlight the urgent need to apply management measures outside the protected area for an effective conservation the Mar Menor lagoon and associated wetlands.

## ACKNOWLEDGEMENTS

The research has been partially supported by the European project DITTY. Development of Information Technology Tools for the Management of European Southern lagoons under the influence of river-basin runoff (EVK3-CT-2002-00084). support which is acknowledged. The research has also been partially supported by the scientific project *Estado ecológico de los humedales mediterráneos semiáridos: propuesta de indicadores para su evaluación* (CGL 2006-08134), funded by the Ministry of Science and Education. whose support is also acknowledged. We also thank Asociación de Naturalistas del Sureste (ANSE) for coordinating, and many anonymous ornithologists for carrying out waterbird censuses in the Mar Menor during the years without public support to this scheme. 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 1995-97 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"

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