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Biodiversity and Conservation of Coastal Lagoons

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1. Introduction

Wetlands are sites of high biodiversity and productivity (Mitsch & Gosselink, 2000). They provide essential services, such as maintenance of atmosphere composition, key habitats for migratory species, and important nursery areas (Basset & Abbiati, 2004), but these ecosystems have suffered a serious decline worldwide due to human influence (Shine & Klemm, 1999; Solimini et al., 2008; Stenert & Maltchik, 2007). Declining water quality, drainage, eutrophication and catchment disturbances such as development, loss of natural vegetation and poor agricultural practices are changing the fundamental ecology of shallow lakes in much of the world (Drake et al., 2011). Different management strategies have been developed to solve these conservation problems, for instance the Europeans Natura 2000 network and Water Framework Directive (WFD).

Among these aquatic systems, coastal wetlands have been subject to massive environmental degradation and habitat destruction worldwide (Goudie, 1990). For example, more than 50% of the original area of coastal wetlands that existed in 1900 has been lost in most countries of Western Europe (Jones & Hughes, 1993). However, not until very recently have they become the focus of conservation interest per se (Abbiati & Basset, 2001; Barnes, 1999) with the declaration of Special Areas of Conservation as a consequence of their listing as a priority habitat type (1150*) on Annex I of the European Union Habitats Directive (Council Directive 92/43/EEC).

Coastal lagoons constitute a common coastal environment, occupying 13% of coastal areas worldwide (Kjerfve, 1994). The conservation of these habitats depends largely on the assessment of their natural characteristics, especially biodiversity, which is one of the main criteria used when elaborating wetland protection policies (Ramsar Convention Bureau, 2005). To assess the conservation status of wetlands correctly it is necessary to include studies of the invertebrate fauna. In this sense, the composition and abundance of benthic invertebrates is one of the most important criteria to be considered. Within this group of organisms are insects, especially Coleoptera and Hemiptera, which are two of the most common groups in these environments and also two of the most important groups in the freshwater food chain.

The purpose of this chapter is to study the importance of the invertebrate fauna in the conservation of coastal lagoons and to assess the effectiveness of protecting areas on the conservation of their biological values. The questions are: "Is the Natura 2000 network

effective in protecting wildlife?" and "Are Coleoptera and Hemiptera assemblages good indicators of the environmental quality of coastal lagoons?" The chapter proceeds with some definitions of "lagoons", and the characterization of the ecology and description of its biological values. Next there is a brief summary of the conservation status of these assemblages within a case study area: coastal lagoons in the Autonomous Region of Galicia, in North-western Spain. The chapter presents results from studies conducted in two periods of time separated by 10 years: 1998 and 2008. The objective was to determine whether the invertebrate fauna has changed in that period and if so, what factors are responsible for that change. The three studied lagoons are catalogued as Special Areas of Conservation (SAC) under the European Union Habitats Directive and two of them are also protected by the Ramsar Agreement. To complement the biological data, several abiotic variables were recorded at the same time as fauna was sampled. The data corresponding to 1998 had already been published in Garrido & Munilla (2008).

The areas where these lagoons are located had suffered an increasing anthropogenic impact in recent years, mainly due to the expansion of tourist areas. Thus it is important to know if the protection figures are effective in the conservation of species and environments. The analysis of the data has allowed us to know the conservation state of the fauna and thus to assess the environmental health of these ecosystems.

2. Definition and characteristics of coastal lagoons

According to the *Interpretation Manual of European Union Habitats* (EUR April 25th, 2003), lagoons are expanses of shallow coastal salt water, of varying salinity and water volume, wholly or partially separated from the sea by sand banks or shingle, or, less frequently, by rocks. Salinity may vary from brackish water to hypersalinity depending on rainfall, evaporation and through the addition of fresh seawater from storms, temporary flooding of the sea in winter or tidal exchange.

Coastal lagoons are ecotones between terrestrial, freshwater and marine ecosystems (Basset & Abbiati, 2004). Sometimes they are mistaken for other coastal inland aquatic ecosystems, such as salt marshes and estuaries (Esteves et al., 2008). Kjerfve (1994) proposed a definition of coastal lagoons that differentiates them from other similar habitat types: an inland water body, usually oriented parallel to the coast, separated from the ocean by a barrier, connected to the ocean by one or more restricted inlets, and having depths which seldom exceed a couple of meters. A lagoon may or may not be subject to tidal mixing, and salinity can vary from that of a coastal fresh-water lake to a hypersaline lagoon, depending on the hydrologic balance. Lagoons formed as a result of rising sea level during the Holocene or Pleistocene and the building of coastal barriers by marine processes. A lagoon evolves from an estuary valley or shallow open embayment to a partially enclosed barrier-lagoon system, and then, with progressive infilling, to a marsh or deltaic-filled lagoon, ending the cycle with a depositional plain or with an eventual destruction by marine erosion (Nichols, 1989).

So, coastal lagoons can be distinguished according to several characteristics: they are close to the coastline, normally closer than one kilometre; they are not completely open to the sea; during the low tide they preserve part of the water isolated from the sea; the water body is separated from the sea and not completely surrounded by dunes; the vegetation does not cover all the wetland surface, leaving open water without submerged vegetation (Soria & Sahuquillo, 2009). Generally speaking, lagoons undergo some important temporal and spatial changes in their abiotic and biological characteristics. These abiotic gradients determine the structure of the biological assemblages (Kjerfve, 1994).

Typical lagoons are water bodies clearly separated from the sea by a sandbar, being their formation related to coastal dynamics. These ecosystems are very heterogeneous in physiography and hydrology (Ponti et al., 2011), due to the varying balance of water, salt, nutrients, particulate organic and inorganic matter (Orfanidis et al., 2008). They can receive freshwater from streams and brooks, runoff or groundwater, as well as seawater by the action of tides or waves. They are characterized by strong directional gradients of salinity, organic matter, nutrients and oxygen concentrations, which act as fine-mesh filters in selecting potential colonizer species (Basset, 2007).

Fig. 1. The Bodeira lagoon, located in the province of Pontevedra (Northwestern Spain).

The hydrological regime is determined by the communication with the open sea and the freshwater inputs. The influence of seawater can be (1) direct through channels, being influenced by tides, or (2) by the action of storms, which makes them unpredictable systems (Casado & Montes, 1995). In this sense, one of the most important characteristics of these ecosystems is the mixture of fresh and seawater, which produces a salinity gradient. According to Kjerfve (1994) coastal lagoons can span the range of salinities from hypersaline to completely fresh. In some cases, a positive vertical gradient of salinity can exist near the outlet if the water is deep enough. This factor is one of the main conditioners of the structure and the assemblages of the communities in these aquatic systems, as well as in the development of many species.

Their depth varies, although it does not exceed the 11 m. (Soria & Sahuquillo, 2009). They are generally shallow, with a high relation between surface and volume. Slope is usually little and the coastal zone is wide, so in many cases all the basin can be considered coastal.

Proximity to the sea makes wind exposure a greater factor than for other inland aquatic ecosystems, resulting in an important effect of vertical mixing and sediment mobilization (Kirk & Lauder, 2000). Primary production is evenly distributed throughout the lagoon without evident depth gradient, except in some special cases where there is a major presence of floating algae.

The organic matter of a lagoon comes from external inputs (runoff or pollution) and the degradation of the autochthonous organic matter. Mineralization of the organic matter can influence the oxygen balance negatively, and excessive inputs of nutrients can result in the eutrophication of the wetland. High levels of organic material cause a decrease in species diversity (Zaldívar et al., 2008).

Lagoons contain quaternary soils formed by the overflow of streams riverbeds. In many instances the action of the vegetation, especially helophytes, leads to the formation of peaty soils. Where the marine influence is greater than the continental input, sandy sediments carried by the wind are frequent. Their relatively low depth and the inputs make filling rates generally high, but these rates vary with the location, because each lagoon differs with respect to freshwater inflow, sediment input, tidal conditions and geomorphic characteristics (Nichols & Allen, 1981). Terrestrial inputs of sediments in the shallowest areas with small slopes reduce the depth and change the physiographical characteristics, which can affect the structure of biological assemblages. This phenomenon depends on the water recharge of the lagoon, the movements of the dunes or the accumulation of the vegetation.

Due to their special situation at the end of a basin, their transitional character (between continental and marine environments), and the interaction with the terrestrial ecosystem, coastal lagoons are high productivity areas (Basset, 2007; Basset et al., 2006a; Esteves et al., 2008; Kjerfve, 1994), but very unstable systems that tend to disappear due to the filling of the basin (Casado & Montes, 1995; Soria & Sahuquillo, 2009). On a geologic time scale, they are short-lived landscape features, in which the environmental transition is geologically rapid and can occur within decades to centuries (Kjerfve & Magill, 1989; Ward & Ashley, 1989).

3. Biodiversity of coastal lagoons

Coastal lagoons support a rich specific biodiversity, so we can find different groups of organisms, like plants, animals or microbes. They act as spawning grounds for marine fish and invertebrates, and behave as resting areas for many species of migratory birds (Aliaume et al., 2007).

Macrophytes can grow almost everywhere in the basin, due to its shallowness and the transparency of the water, except in more unstable or seasonally dry areas. Only nutrients determine the vegetation growth and the turbidity of the water by phytoplankton in cases of overconcentration. Lagoon vegetation constitutes a particular habitat for many species, especially birds and invertebrates, and is the substrate on which periphytic communities develop. They provide refuge, food and substrate for aquatic organisms (Diehl & Kornijów, 1998). Moreover, macrophytes create a micro climate with softened temperatures during the hot summer, produce oxygen and retain the upper horizon of the sediment helping to control the turbidity. In coastal lagoons the helophytes usually cover a more or less wide stretch in the shallowest areas, acting as a buffer zone for impacts on the open water. Submerged macrophytes can reduce eutrophication by stabilizing clear water states or outcompeting phytoplankton for nutrients and light (Søndergaard & Moss, 1998). But they

can have a negative contribution to the productivity in most shallow lakes, because they influence water exchange with the offshore area, reducing water movement and increasing sedimentation rates (Arocena, 2007).

Fig. 2. Proliferation of the water lily (*Nymphaea alba*) in the Xuño lagoon (A Coruña, Northwestern Spain).

The microbial community is composed of common heterotrophic taxa of natural waters, the presence of coliforms in habitats suffering from human impact being of special interest (Soria & Sahuquillo, 2009). The bacterial community is basic in the energy flows, because it decomposes and remineralizates organic matter. The hydrology of the lagoon (freshwater or marine inputs) greatly influences the bacterial composition (Piccini & Conde, 2004). Changes in the ecosystem functioning can change the composition of the bacterial assemblages as an answer to the stress factor (Langenheder et al., 2003).

Phytoplankton in coastal lagoons is rich and diverse (Ramil et al., 2007), and as primary producers are an important part of the food web in freshwater environments. Phytoplankton is usually composed of diatoms, dinoflagellates, chlorophytes, cryptophytes and other microflagellates. In the coastal lagoons it is often responsible for the eutrophication phenomenon, with dominance of phytoplankton over other groups of organisms.

Zooplankton consists of heterotrophs that live suspended in the water column, and includes protists (flagellates and ciliates) and micro animals, mainly rotifers and microcrustaceans (cladocerans, copepods and ostracods). Overall, the plankton is consumed by fish, macroinvertebrates and some waterfowl, especially in brackish systems.

Benthos comprise the set of mobile or sessile aquatic organisms living in or on the substrate. They are very important in the recycling of the organic matter, facilitating the activity of the bacterial decomposers (Casado & Montes, 1995). With regard to photosynthetic organisms, besides being part of the plankton communities of microalgae (phytoplankton), many microscopic primary producers can grow attached to solid substrates, on rocks (epilithon), on the sediment (episammon) or on the vegetation (periphyton).

Zoobenthos are macroinvertebrates that live on the bottom (benthos)of water bodies, and on some macrophytes. In addition to the larvae of numerous species of insects (especially flies, dragonflies, beetles and bugs), zoobenthos includes flatworms, annelids, molluscs and crustaceans. In lentic ecosystems, zoobenthos is very important compared with the pelagic communities of animals, particularly in shallow systems and coastal areas. Surface area, water salinity, and outlet width and length, can actually be considered the key limiting, dimensions defining the environmental niche space for benthic macroinvertebrates in lagoon ecosystems (Basset et al., 2006b).

Fish are the main organisms of nekton, i. e. all the organisms that actively swim in the water. In coastal lagoons, the presence of fish is common and is related to the salinity of the system and its opening to the sea. Lentic ecosystems are an ideal habitat for waterfowl and amphibians. Nowadays amphibians are in decline worldwide, partly due to the decline of aquatic ecosystems where they live, largely coastal lagoons. On the other hand, waterfowl are important consumers in lagoons, but can also be sources of nutrients with a negative effect on eutrophication. Besides birds, amphibians and fishes, other vertebrates such as certain species of reptiles and mammals are typical of these ecosystems, although the diversity of these animals is more limited and only a few species are related to coastal lagoons.

4. Hazards and conservation management

Modification of extensive natural areas by human action, such as desiccation, urbanization, hydrologic modification and isolation of areas previously connected, has led to a reduction or disappearance of large areas of wetlands and coastal lagoons. Increasing of tourism pressure and the use of these ecosystems for different human activities such as aquaculture or fisheries has induced internal perturbations such as pollution or removal of indigenous species (Aliaume et al., 2007).

In coastal lagoons, species distribution is affected by heterogeneity of the environmental variable, such as water depth, hydrodynamic conditions, sediment characteristics, as well as by sources of anthropogenic disturbance that could interact with natural heterogeneity affecting patterns of species distribution, species diversity and ecosystem functioning (Ponti et al., 2011). Sometimes it is difficult to distinguish between natural heterogeneity and human disturbance.

Healthy lagoons produce food for birds and fish, and provide for several human extractive activities and significant food resources (aquaculture, fishing, hunting, etc.) (Sylaios & Theocharis, 2002). However, an increase in primary production by external inputs of nutrients, mainly anthropogenic, stresses the ecosystem favouring opportunistic species and reducing diversity. Unfortunately, coastal areas are highly impacted by eutrophication (Nixon, 1995).

These systems are very sensitive to potential climate change due to their dependence on continental and marine inputs (Álvarez-Cobelas et al., 2005). Recent climate changes are

beginning to have effects on these environments (Soria & Sahuquillo, 2009). According to the Assessment Report of the Intergovernmental Panel on Climate Change (IPCC) (2007), many observed changes in freshwater biological systems are associated with rising water temperatures, as well as related changes in salinity, oxygen levels and circulation. Sea level rise and human development are together contributing to losses of coastal wetlands and increasing damage from coastal flooding in many areas. Climate change is likely to reduce flooded areas and increase eutrophication, increasing helophytes populations and grasslands and reducing submerged macrophytes.

One of the major consequences of the climate change is the spread of exotic species worldwide (Ricciardi, 2006), affecting the ecological functions of an ecosystem (Maezono & Miyashita, 2003; Ricciardi et al., 1997) and causing a loss of indigenous biodiversity (Rodríguez-Pérez et al., 2009; Witte et al., 2000).

It will also affect the biogeochemical cycles of coastal lagoons, especially in peaty environments, characterized by low concentrations of nutrients and organic matter accumulation. In this sense, the relative sedimentation ratio is extremely important in the assessment of lagoon behaviour under climate change scenarios, particularly regarding the prospects for accelerated sea level rise. Research into sedimentation rates is essential to formulating management plans for lagoons under present conditions of sea level rise, as it is to making an informed assessment of the hazards that might be posed by climate change (Kirk & Lauder, 2000).

5. Case study

5.1 Introduction

Major gaps still remain in aquatic conservation knowledge of lagoons and other coastal brackish water habitats in temperate regions, and much more emphasis needs to be placed on the investigation of the ecology of such systems to ensure their effective management (Barnes, 1999). Information about the relative biodiversity value of different water body types is a vital pre-requisite for many strategic conservation goals (Williams et al., 2003), including sustainable catchment management as required by the EC Water Framework Directive (2000/60/EC). In this sense, the composition and abundance of benthic invertebrates is one of the most important criteria to be considered.

Little is known of the communities of aquatic insects in coastal lagoons, especially when compared with freshwater ecosystems (Garrido & Munilla, 2008). Within this group of organisms are Coleoptera and Hemiptera, two of the most common groups in these environments. Water beetles are generally considered a suitable group to assess the environmental and conservation value of wetland sites and habitats (Bilton et al., 2006; Davis et al., 1987; Eyre & Rushton, 1989; Eyre et al., 1993; Foster, 1987, 1999; Foster et al., 1990; Oertli et al., 2005; Pérez-Bilbao & Garrido, 2009; Ribera & Foster, 1992; Sánchez-Fernández et al., 2006). The detailed study of the autoecology of aquatic insects, especially water beetles, has revealed the existence of many species with narrow ecological requirements including adaptations to saline environments (Foster, 2000; Greenwood & Wood, 2002). Aquatic Hemipteran assemblages are generally poorer in species than are water beetles, and seem to be more resilient to environmental change (Broering & Niedringhaus, 1988; Eyre & Foster, 1989; Roback, 1974; Savage, 1996; Tuly et al., 1991; Vierssen & Verhoeven, 1983). However, several families display differing tolerances to water pollution and are potential bioindicators of water quality (Jansson, 1977), so they can

be used in terms of regional or global conservation planning of freshwater biodiversity (Polhemus & Polhemus, 2008).

The main objectives of this study were (a) to analyse the composition and structure of aquatic Coleoptera and Hemiptera assemblages in three protected coastal lagoons using data obtained in studies carried out in two periods of time separated by 10 years (1998 and 2008); (b) to assess the importance of the invertebrate fauna (Coleoptera and Hemiptera) in the conservation of these ecosystems; (c) and to assess the effectiveness of protecting areas (Natura 2000) on the conservation of their biological values studying the change in these assemblages after 10 years.

Fig. 3. The Vixán lagoon, located in the province of A Coruña (Northwestern Spain).

5.2 Study area

The study area comprised three coastal lagoons in the Autonomous Region of Galicia (North-western Spain): Bodeira, Xuño and Vixán (Figures 1-4). The three lagoons are located within Special Areas of Conservation (SAC) under the European Union Habitats Directive (Directive 92/43/EEC). Vixán and Bodeira are also protected by the Ramsar Agreement (Ramsar sites no. 598 and 452, respectively). Although each of the lagoons selected was completely separated from the sea by a barrier of sand dunes, seawater can enter by filtration or when sea storms break through the sand barrier. Freshwater is provided by small streams, rainfall and runoff. The extent of the standing water ranged from 0.8 Ha in Bodeira, to 11.2 Ha in Vixán and 2.3 Ha in Xuño. According to the thermal classification of lakes proposed by Lewis (1983), these lagoons are warm polymictic, without a stable thermal stratification. Bodeira and Xuño are fresh water systems, and Vixán is considered an oligohaline water body (Ramil et al., 2007).

Vixán has a relatively large reed (*Phragmites australis* (Cav.) Trin. Ex Steud) bed. Bodeira presents species such as *Polygonum amphibium* L., *Glyceria fluitans* (L.) R. Br., *Ranunculus* spp. L or *Myriophyllum* spp. Xuño has an important assemblage of the water lily (*Nymphaea alba*

L.) and also has different species of macrophytes including *Hydrocotyle vulgaris* L., *Ranunculus* spp. or *Veronica* spp.

The climate of the study area is mild and wet, with annual average temperatures around 14°C and annual rainfall between 1200 and 1800 mm. The landscape consists of a mosaic of farmland, heather (*Erica* spp.) and gorse (*Ulex* spp.) heathlands and pine (*Pinus pinaster* Ait.) forests in a densely populated rural setting (Garrido & Munilla, 2008).
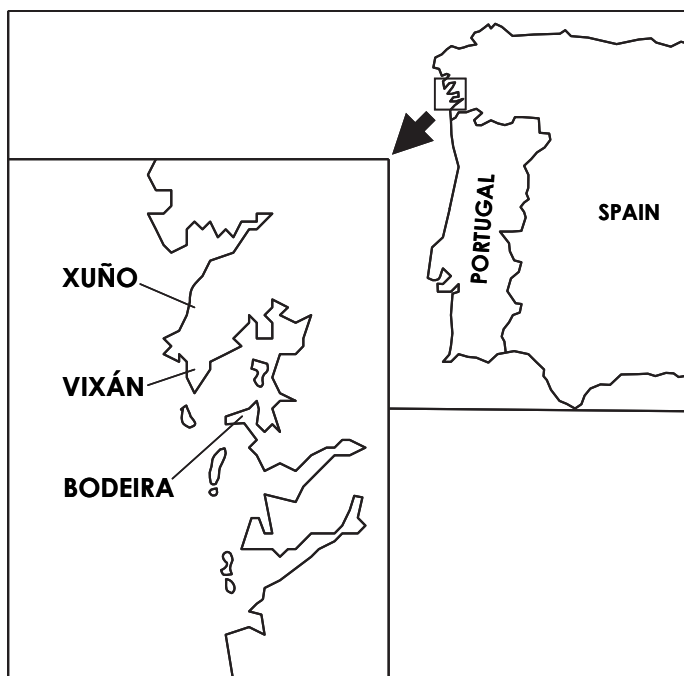

Fig. 4. Map of the study area showing the location of the lagoons under study (based on Garrido & Munilla, 2008).

5.3 Material and methods
5.3.1 Sampling methods and variables measured

The three lagoons were sampled in spring and summer in two periods, 1998 and 2008. The fauna was collected by sweeping an entomological water net (500 µm mesh, 30 cm diameter and 60 cm deep) across a 10m transect running parallel to the margin. This semi-quantitative method allows for direct comparisons across sites or time because sampling effort can be assumed equivalent. The material was preserved in 99% ethanol, and sorted out and identified to species level at the laboratory. After being studied, they were conserved in 70% ethanol and deposited in the scientific collection of the Entomology Laboratory at Vigo University. Water temperature, pH, conductivity, and dissolved oxygen were measured at each lagoon at the same time as the fauna was sampled. Salinity was only measured in 1998. As the salinity and conductivity values were highly correlated (r= 0.994) (Garrido & Munilla, 2008) and as this variable provides redundant information we decided not to measure it in 2008.

5.3.2 Data analysis

The structure of the assemblage was assessed using different diversity indices: Species richness (S); Rarefied species richness (ES); Abundance (N) and the Shannon-Wiener diversity index (H'). The values of rarefied richness were calculated for 100 individuals ES (100). ES and H' were calculated using the program PRIMER version 6. Analysis of variance (two-way ANOVA) was used to test for significant differences between the three lagoons and the two periods of time in both diversity indices and environmental variables. ANOVA was run using SPSS version 19.

Relationships between environmental variables with diversity indices and species were determined by a Pearson correlation test. Prior to this, the Kolmogorov-Smirnov test was used to verify the normal distribution of the data. Variables not showing normal distribution were logarithmically transformed ($\log_{10}$). These analyses were performed using SPSS version 19.

Complete linkage cluster analysis with Bray-Curtis coefficient was used to cluster the samples into groups and thus to be able to verify the changes in the Coleoptera and Hemiptera assemblages composition. This analysis was carried out with PRIMER version 6.

Canonical correspondence analysis (CCA) was used to analyze species-environment relationships in order to identify environmental factors potentially influencing water beetle assemblages. A Monte Carlo permutation test was performed to assess the significance of the relationships between environmental variables and species composition among study sites (Heino, 2000). CCA was carried out on global abundances, that is, total number of individuals collected at a site over the sampling period. Species with less than 10 individuals were removed from the analysis, which was performed using the CANOCO 4.5 program (Ter Braak & Šmilauer, 2002). The environmental factors used were pH, water temperature, dissolved oxygen and conductivity.

5.4 Results

In total, 71 species of 11 families of aquatic Coleoptera (55 species) and 8 families of aquatic Hemiptera (16 species) were collected (Table 1). The most representative families of beetles were Dytiscidae (20 species), Hydrophilidae (13 species), Haliplidae and Hydraenidae (5 species) and among bugs the most representative families were Corixidae (5 species) and Notonectidae (4 species).

Only 36 species (27 Coleoptera and 9 Hemiptera) were recorded in both periods, corresponding to approximately half the total species recorded (71). Besides, 20 species (15 Coleoptera and 5 Hemiptera) were only recorded in 1998 and 15 species (13 Coleoptera and 2 Hemiptera) were only recorded in 2008.

5.4.1 Biogeography

According to the classification proposed by Ribera et al. (1999) for Coleoptera and Millán et al. (2002) for Hemiptera, the species were classified in their respective chorological categories. In the three studied lagoons, species of four categories were captured: northern, trans-Iberian, southern and endemic (Figure 5). In the two years, most of the species belonged to the trans-Iberian category (84-88%), followed by northern (5-10%), endemics (4-5%) and southern (2%). It has to be noted that all the species of aquatic bugs are trans-Iberian, i. e. they are present in Europe north of Pyrenees, the Iberian Peninsula and north Africa (Table 1).
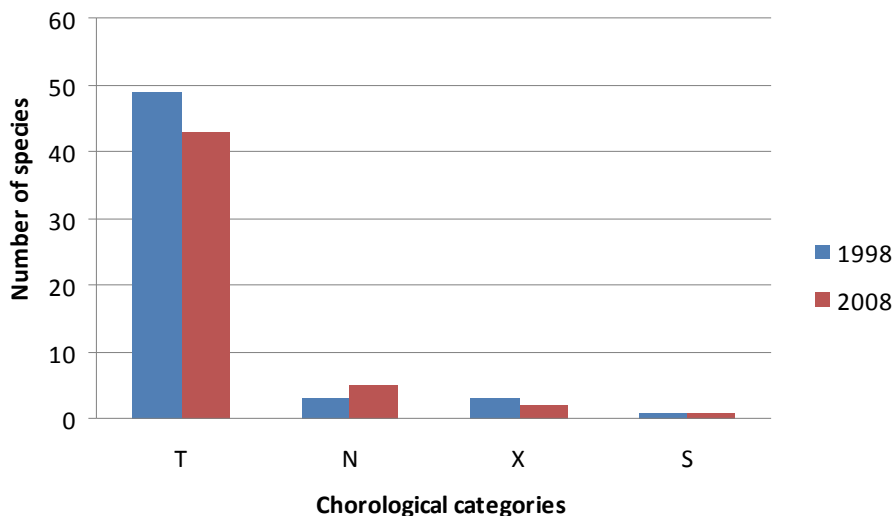
Fig. 5. Species composition of the chorological categories of aquatic Coleoptera and Hemiptera captured in the two periods of time. S: southern; X: endemic; N: northern; T: Trans-Iberian.

5.4.2 Abundance, species richness and diversity

The total abundance was of 10,166 individuals (4,395 Coleoptera and 3,904 Hemiptera). In 1998, 3,399 individuals were captured (1,005 beetles and 2,394 bugs), and in 2008 were 6,767 (3,390 Coleoptera and 3,377 Hemiptera). The greatest abundance was observed in the Bodeira lagoon in 2008, with 3,732 individuals collected (1,856 in spring and 1,876 in summer). On the contrary, the lowest value was obtained in Vixán in spring of 2008 with 206 individuals. In 1998 the hemipteran families with the highest number of individuals were Corixidae (2,032) and Pleidae (285). Among coleopterans, we found the highest abundances for Dytiscidae (381), Hydrophilidae (272) and Hydraenidae (180). On the contrary, in 2008 the hemipteran families with the highest number of individuals were Pleidae (2,533) and Corixidae (240). Among coleopterans, we found the highest abundances for Hydraenidae (2,045), followed by Hydrophilidae (670) and Dytiscidae (380).

In 1998 the most abundant species of aquatic bug found in the study area was *Hesperocorixa linnaei* (1,146), while in 2008 it was *Plea minutissima* (2,533). Within water beetles, the most abundant species in 1998 was *Hygrotus inaequalis* (202 individuals) and in 2008 it was *Ochthebius viridis fallaciosus* (1,812 individuals).

Only one species of Coleoptera (*Hygrotus inaequalis*) and one of Hemiptera (*Plea minutissima*) were recorded in all samples, in the three lagoons and in the two periods, which corresponds to only 1.42% of species. On the other hand, 10 species of water beetles *(Haliplus heydeni, H. guttatus, Hydroglyphus geminus, Hydroporus vespertinus, Graptodytes varius, Dytiscus marginalis, Helophorus flavipes, Hydrochus flavipennis, Berosus signaticollis* and *Dryops striatellus)* and 4 of bugs (*Nepa cinerea, Hesperocorixa sahlbergi, Notonecta glauca* and *Gerris gibbifer*) were recorded in just one of the samples, which corresponds to 19.72% of species.

Species	Bodeira 1998	Bodeira 2008	Vixán 1998	Vixán 2008	Xuño 1998	Xuño 2008	Distribution
COLEOPTERA							
Gyrinidae							
Gyrinus (*Gyrinus*) *caspius* Ménétriés, 1832		1	4		4	7	T
Haliplidae							
Peltodytes caesus (Duftschmid, 1805)	2		8	2	1	13	T
Peltodytes rotundatus (Aubé, 1836)		2	1				T
Haliplus (*Neohaliplus*) *lineatocollis* (Marshan, 1802)		2	4				T
Haliplus (*Haliplus*) *heydeni* Wehncke, 1875			1				T
Haliplus (*Liaphlus*) *guttatus* Aubé, 1836						2	T
Noteridae							
Noterus laevis Sturm, 1834	5	22	8	13		116	T
Paelobiidae							
Hygrobia hermanni (Fabricius, 1775)	6	5			51	3	T
Dytiscidae							
Liopterus haemorrhoidalis (Fabricius, 1787)		1	1				T
Laccophilus minutus (Linnaeus, 1758)	1	5	3	2	2	66	T
Hyphydrus aubei Ganglbauer, 1891	1	8	4	1	9	9	T
Hydrovatus clypealis Sharp, 1876		5		1		16	T
Bidessus goudoti (Laporte, 1835)	27	23	4		41	87	T
Hydroglyphus geminus (Fabricius, 1792)			1				T
Hygrotus inaequalis (Fabricius, 1777)	78	56	100	17	24	6	T
Hygrotus lagari (Fery, 1992)						2	S
Hydroporus vagepictus Fairmaire & Laboulbène, 1854					2	1	X
Hydroporus vespertinus Fery & Hendrich, 1988	1						X
Graptodytes aequalis Zimmermann, 1918			4				S
Graptodytes flavipes (Olivier, 1795)	1		14				T
Graptodytes ignotus (Mulsant, 1861)			2				T
Graptodytes varius (Aubé, 1838)						11	T
Stictonectes lepidus (Olivier, 1795)			3				T
Agabus bipustulatus (Linnaeus, 1767)		1	1		1		T
Rhantus (*Rhantus*) *suturalis* (McLeay, 1825)		2	47	8	4	1	T
Colymbetes fuscus (Linnaeus, 1758)		1					T
Dytiscus marginalis Linnaeus, 1758			1				T
Cybister (*Scaphinectes*) *lateralimarginalis* (De Geer, 1774)			3		1		T
Helophoridae							
Helophorus (*Rhopalhelophorus*) *flavipes* Fabricius, 1792				1			N
Helophorus (*Rhopalhelophorus*) *minutus* Fabricius, 1775	8	2	55	5	2	2	N
Hydrochidae							
Hydrochus angusi Valladares, 1988			1	7			X
Hydrochus angustatus Germar, 1824			1			26	T
Hydrochus flavipennis Küster, 1852						6	T
Hydrophilidae							
Berosus (*Berosus*) *affinis* Brullé, 1835	28		8		83	19	T
Berosus (*Berosus*) *hispanicus* Küster, 1847			57		2		T
Berosus (*Berosus*) *signaticollis* (Charpentier, 1825)						9	T
Paracymus scutellaris (Rosenhauer, 1856)		8		3		6	T
Anacaena lutescens (Stephens, 1829)			49	19	2	44	T

Table 1. Aquatic Coleoptera and Hemiptera species recorded in the lagoons Bodeira, Vixán and Xuño in the years 1998 and 2008, with their abundance and the biogeographical distribution.

	1	2	3	4	5	6	Dist.
Helochares (*Helochares*) *lividus* (Forster, 1771)	4				5		T
Helochares (*Helochares*) *punctatus* Sharp, 1869		149				79	N
Enochrus (*Lumetus*) *fuscipennis* (Thomson, 1884)		2		19		182	T
Enochrus (*Lumetus*) *halophilus* (Bedel, 1878)	1		7		1		T
Cymbiodyta marginella (Fabricius, 1792)			9	6		27	N
Hydrobius fuscipes (Linnaeus, 1758)		4				1	T
Limnoxenus niger (Gmelin, 1790)		25	12	29	1	39	N
Hydrophilus (*Hydrophilus*) *pistaceus* Laporte, 1840			2		1		T
Hydraenidae							
Hydraena testacea Curtis, 1830				4	1	14	T
Limnebius furcatus Baudi, 1872		26	27	6		182	T
Ochthebius (*Asiobates*) *dilatatus* Stephens, 1829			1	1			T
Ochthebius (*Ochthebius*) *nanus* Stephens, 1829	1		1				T
Ochthebius (*Ochthebius*) *viridis fallaciosus* Ganglbauer, 1901		851	147	25	2	936	T
Elmidae							
Oulimnius rivularis (Rosenhauer, 1856)		12	1	2			T
Dryopidae							
Dryops algiricus (Lucas, 1849)		78	4			8	T
Dryops luridus (Erichson, 1847)				2		6	T
Dryops striatellus (Fairmaire & Brisout, 1859)					5		T
HEMIPTERA							
Nepidae							
Nepa cinerea Linnaeus, 1758		1					T
Ranatra linearis (Linnaeus, 1758)	1			1			T
Corixidae							
Corixa panzeri (Fieber, 1848)	221	13	26	13	160		T
Hesperocorixa linnaei (Fieber, 1848)	385	91	29	50	732	73	T
Hesperocorixa sahlbergi (Fieber, 1837)			4				T
Sigara (*Halicorixa*) *stagnalis stagnalis* (Leach, 1817)	16		150		288		T
Sigara (*Subsigara*) *scotti* Douglas & Scott, 1868	20				1		T
Naucoridae							
Naucoris maculatus maculatus Fabricius, 1798	17	336	12	41	16	20	T
Notonectidae							
Anisops sardeus sardeus Herrich-Schäffer, 1849		153		8		1	T
Notonecta (*Notonecta*) *glauca glauca* Linnaeus, 1758			2				T
Notonecta (*Notonecta*) *meridionalis* Poisson, 1926			21			2	T
Notonecta (*Notonecta*) *viridis* Delcourt, 1909	1	4		2	4	1	T
Pleidae							
Plea minutissima minutissima Leach, 1817	238	1812	33	536	14	185	T
Hydrometridae							
Hydrometra stagnorum (Linnaeus, 1758)		12	1	1			T
Veliidae							
Microvelia (*Microvelia*) *pygmaea* (Dufour, 1833)		19	1	2			T
Gerridae							
Gerris (*Gerris*) *gibbifer* Schummel, 1832			1				T
Total abundance	1063	3732	876	827	1460	2208	

Table 1. (continued). Aquatic Coleoptera and Hemiptera species recorded in the lagoons Bodeira, Vixán and Xuño in the years 1998 and 2008, with their abundance and the biogeographical distribution.

In total, the species richness of the two periods of time (1998 and 2008) was of 71 species. In 1998, 56 species (42 Coleoptera and 14 Hemiptera) were captured, and 51 in 2008 (40 beetles and 11 bugs). If we analyse these results by lagoons, we can see that in 1998 Vixán presented the highest value of water beetle richness with 36 species, followed by Xuño (22) and Bodeira (14). On the other hand, ten years later it is Xuño that presents the highest richness with 31 species, followed by Bodeira (24) and Vixán (21). Aquatic bug richness did not vary too much between lagoons and in 1998 was again Vixán the richest one (11), followed by Bodeira (8) and Xuño (7). In 2008, both Bodeira and Vixán presented the same richness value with 9 species, and Xuño was still the poorest lagoon (6).

Rarefied species richness is the expected number of species for a given number of randomly sampled individuals and facilitates comparison of areas in which densities may differ (McCabe & Gotelli, 2000). The Bodeira lagoon had the lowest values in summer of 2008 (7.65) and in summer of 1998 (7.71). The highest values correspond to Vixán in spring of 1998 (23.75) and to Xuño in summer of 2008 (18.86).

Diversity indices	1998			2008		
	Mean ± SD	Minimum	Maximum	Mean ± SD	Minimum	Maximum
Richness S	22.17 ± 9.02	12	38	24.83 ± 3.97	20	30
Rarefied Richness ES (100)	12.57 ± 5.92	7.71	23.75	13.48 ± 3.93	7.65	18.86
Abundance N	566.50 ± 163.14	395	782	1127.83 ± 689.60	206	1876
Diversity H'(log2)	2.65 ± 0.83	1.95	4.18	2.46 ± 0.71	1.67	3.77

Table 2. Mean, minimum and maximum values of the diversity indices measured in 1998 and 2008.

The Shannon-Wiener index (H'(log2)) revealed that, in general, the studied lagoons presented high diversity values. The lowest diversity was recorded in Bodeira in summer of 2008 (1.67) and the highest in Vixán in spring of 1998 (4.18). The mean diversity values between the two periods of time were very similar. In 1998, the mean diversity was 2.65, while in 2008 it was 2.46.

Table 2 shows the mean, minimum and maximum values of the diversity indices for the coastal lagoons studied. There were no significant differences ($p < 0.05$) among periods (1998 and 2008) in any of the diversity indices, as evidenced by the ANOVA (Table 3).

Diversity indices and environmental variables	ANOVA Factor 1		ANOVA Factor 2	
	F	p	F	p
Richness S	0.439	0.522	1.078	0.38
Rarefied Richness ES (100)	0.099	0.759	2.469	0.14
Abundance N	3.765	0.081	0.981	0.412
Diversity H'(log2)	0.195	0.668	3.488	0.076
Water Temperature (°C)	0.006	0.941	0.64	0.55
pH	3.172	0.105	2.071	0.182
Conductivity (µS/cm)	0.989	0.343	1.166	0.355
Dissolved Oxygen (mg/l)	0.097	0.762	0.599	0.57

Table 3. Results of the ANOVA with years (1) and lagoons (2) as factors.

5.4.3 Environmental variables

Table 4 shows mean, minimum and maximum values of the environmental variables measured in the two years. The main result to highlight is the extreme value of conductivity measured in the summer of 1998 in the Vixán lagoon (34,816.60 μS/cm), much higher than in the other surveys. However, the ANOVA showed no significant differences ($p < 0.05$) among periods (1998 and 2008) in any of the environmental variables (Table 3).

Environmental variables	1998			2008		
	Mean ± SD	Minimum	Maximum	Mean ± SD	Minimum	Maximum
Water Temperature (°C)	21.45 ± 1.70	19.45	24.35	21.33 ± 3.33	16.40	25.60
pH	7.72 ± 0.92	6.70	8.95	7 ± 0.35	6.65	7.65
Conductivity (μS/cm)	6,296.55 ± 13,975.86	263.45	34,816.60	618.67 ± 443. 88	227	1,408
Dissolved Oxygen (mg/l)	8.75 ± 4.85	2.45	14.20	7.79 ± 5.85	2.03	16.50

Table 4. Mean, minimum and maximum values of the environmental variables measured in 1998 and 2008.

5.4.4 Correlations

The Pearson correlation test was performed to assess the relation between the environmental variables and the species and diversity indices. There were no significant correlations (p<0.05) between physicochemical variables and the ecological parameters. However, we found several significant and positive correlations between these variables and species. The species *Gyrinus caspius* was correlated with temperature (r= 0.66); *Corixa panzeri* (r= 0.82) and *Sigara scotti* (r= 0.65) were related to pH, while *Berosus hispanicus* (r= 1.00) and *Rhantus suturalis* (r= 0.92) presented high correlations with conductivity.

5.4.5 Influence of environmental variables on Coleoptera and Hemiptera assemblages

Figure 6 shows the results of the CCA. The eigenvalues for axes 1-4 were 0.466, 0.282, 0.226 and 0.076 respectively. Correlations for axes III and IV with environmental variables were low (r < 0.5), and only axes I and II were used for data interpretation. The cumulative percentage of variance for the species-environmental relation for these two axes was of 71.2%. The first two canonical axes were significant, as shown by the Monte Carlo permutation test (p = 0.004).

The first principal gradient is positively correlated with oxygen (r = 0.453) and temperature (r = 0.442), and negatively correlated with pH (r = -0.585). The second gradient is positively correlated with conductivity (r = 0.621) and negatively correlated with oxygen (r = -0.779), temperature (r = -0.642) and pH (r = -0.447).

According to the diagram, the species *Berosus hispanicus, Rhantus suturalis* and *Sigara stagnalis* are related to sites with high values of conductivity, while species such as *Sigara scotti* or *Corixa panzeri* prefer sites with high values of pH.
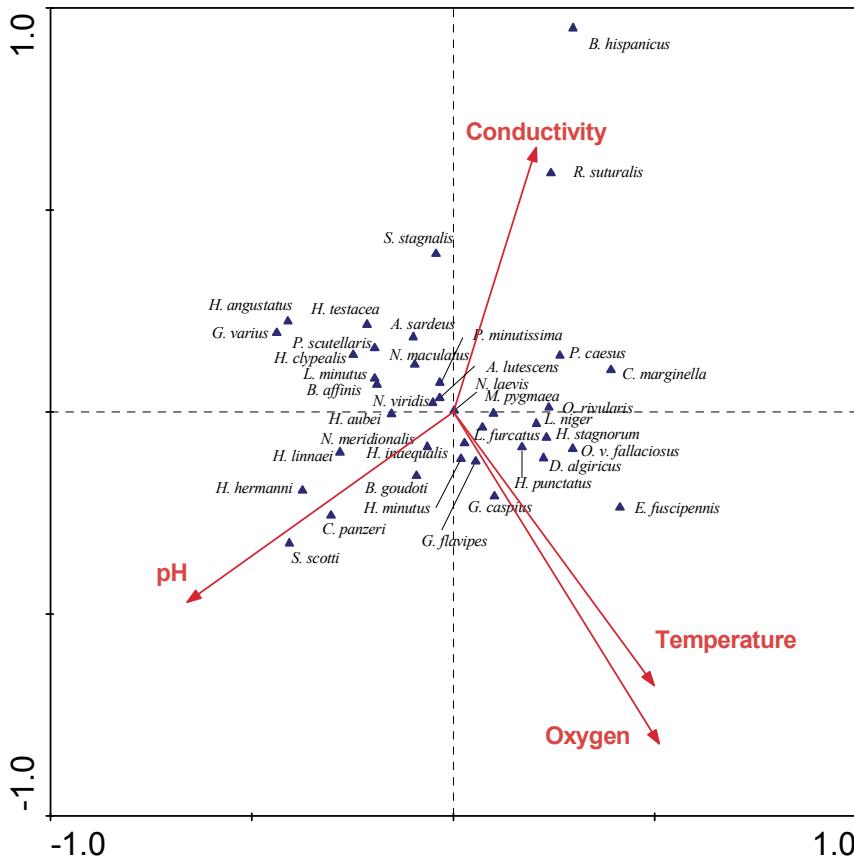
Fig. 6. Results of the canonical correspondence analysis (CCA) based on the aquatic Coleoptera and Hemiptera assemblages with respect to environmental variables. Arrows represent the environmental variables and triangles the species.

5.4.6 Assemblage composition

In 1998, 10 species were exclusive (not recorded in 2008) in Bodeira (7 Coleoptera and 3 Hemiptera), 26 in Vixán (21 Coleoptera and 5 Hemiptera) and 11 in Xuño (8 Coleoptera and 3 Hemiptera). On the other hand, in 2008, 22 species were exclusive (not recorded in 1998) in Bodeira (18 Coleoptera and 4 Hemiptera), 9 in Vixán (6 Coleoptera and 3 Hemiptera) and 18 in Xuño (16 Coleoptera and 2 Hemiptera).

The Bray-Curtis coefficient was used to calculate the affinity between samples, because it allows the comparison between the assemblages of different ecosystems or in the same ecosystem at different moments. The results ranged from 0 to 1. If the value obtained is close to 1, the populations will be more similar. In this study twelve samples were used, six in 1998 and six in 2008. The greatest degree of affinity was observed between spring and summer of 1998 in Bodeira (64.53%) and between Bodeira and Xuño in spring of 2008 (61.41%). If we compare the two periods of time, we observe a low degree of affinity in the

three studied lagoons (Xuño: 10.79; Bodeira: 18.89; and Vixán: 23.84). In the figure 7, the temporal affinity is greater than the spatial one, i. e. the studied lagoons were more similar between them in the same year than the same lagoon in the two different periods of time. Two main groups corresponding to 1998 and 2008 are represented. In the "1998" group, it is important to note that Vixán is isolated because it had a low affinity with other lagoons (18%).

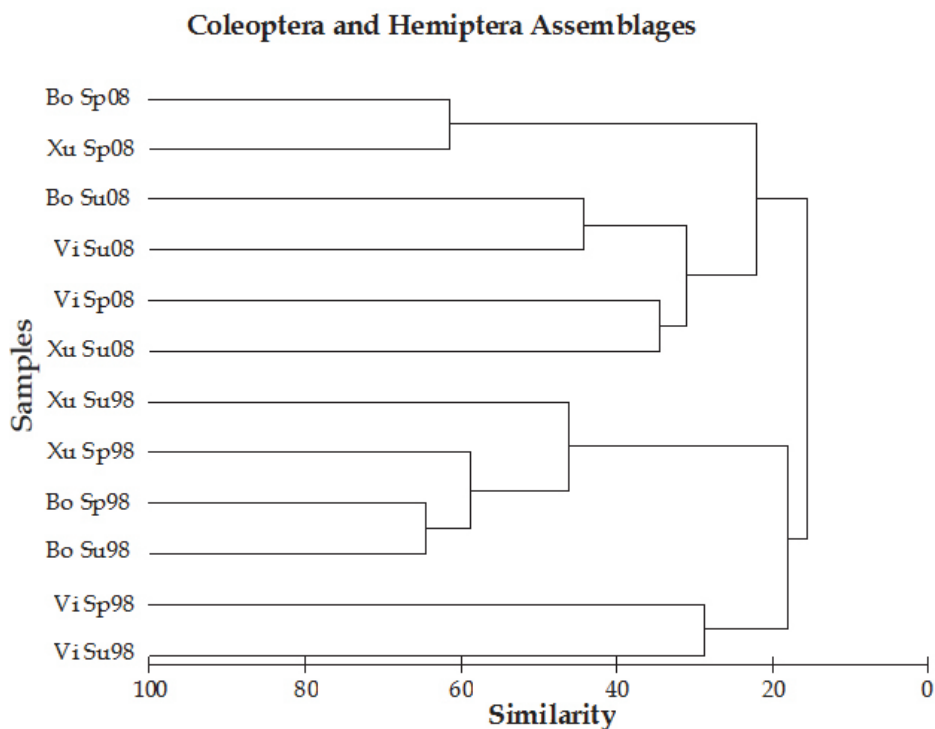### Coleoptera and Hemiptera Assemblages

Fig. 7. Dendrogram from complete linkage clustering based on the Bray-Curtis coefficient.

6. Discussion

The aquatic Coleoptera and Hemiptera assemblages in the studied lagoons were predominantly composed of species that are widely distributed on the regional scale. Most of the species of water beetle collected had a wide Palearctic distribution and only three species could be considered endemic to Iberia: *Hydroporus vagepictus*, *Hydroporus vespertinus* and *Hydrochus angusi* (Ribera et al., 1999; Ribera & Vogler, 2004). The paucity of endemics among coastal water beetles has been attributed to the temporal dynamics of the lagoons and their recent origin in geological time, as most of the sites are 'temporary' on a scale of a few hundred years (Ribera et al., 1996) and this could also apply to aquatic Hemiptera. According to Ribera & Vogler (2000), stagnant water bodies present less endemic species than running waters, because stagnant species have a greater dispersal capability and thus a greater geographical distribution.

It should be noted that Vixán is one of the few localities where *Hydrochus angusi* (Pérez-Bilbao et al., 2009) and *Cymbiodyta marginella* (Pérez-Bilbao et al., 2010; Sáinz-Cantero & Garrido, 1996) have been recorded in the Iberian Peninsula. Regarding water bugs, we can highlight the species *Sigara scotti* and *Notonecta glauca glauca*, which are distributed in the Iberian Peninsula as a few scattered populations (Nieser & Montes, 1984; Nieser et al., 1994).

In the three lagoons, Coleoptera assemblages were less similar in species composition across sites than Hemiptera assemblages. Compared with other coastal stagnant water bodies in the Iberian Peninsula and the Balearic Islands, species richness of these coastal lagoons can be considered high. It is important to highlight that only three lagoons were sampled, when in other studies the number of sampling sites is much higher, e. g. 13 coastal wetlands in Ribera et al. (1996), 9 sites in Garrido et al. (1996, 1997), 22 ponds in Florencio et al. (2009) or 38 lagoons in Lucena-Moya et al. (2010). A richer Coleoptera assemblage seems to be the rule in aquatic insect communities where the proportion of Coleoptera to Hemiptera usually varies from 3:1 to 4:1 (Eyre & Foster, 1989; Lancaster & Scudder, 1987; Millán et al., 1997, 2001; Moreno et al., 1997). This lends further support to the idea that, for coastal brackish-water systems, Coleoptera assemblages are more valuable than Hemiptera assemblages for characterizing water bodies and providing information about their conservation status (Eyre & Foster, 1989; but see Hufnagel et al., 1999). In habitats with high conductivity values this proportion (3:1 to 4:1) can indicate a good conservation state (Sánchez-Fernández et al., 2007). On the other hand, fewer similarities in species composition among sites imply that a larger proportion of wetland sites will be needed to ensure the protection of entire assemblages (Margules et al., 1988).

According to Duigan et al. (1998), species richness is one of the most important parameters when assessing the conservation state of an ecosystem. The aquatic assemblages of the studied coastal lagoons seem to reflect adequate conditions for the conservation of biodiversity in terms of species richness. One aspect to consider in the conservation and management of aquatic ecosystems, apart from the danger of extinction, isolation and fragmentation, is the rarity of the species in terms of its distribution range, habitat specificity and abundance of their populations (Gaston, 1994). The existing information on habitat specificity of each species is not abundant in general and this is due to the lack of autoecological studies, because abundance data are often estimates, or simply indicate the presence/absence of species (Millán et al., 2002).

The observed values of physicochemical variables were within the range considered normal for these types of environments (fresh water and oligohaline water systems) in both periods (1998 and 2008). These values were similar to those obtained by other authors (Boix et al. 2008; Della Bella et al. 2008), except Vixán in summer 1998, probably due the entrance of sea water caused by a storm.

The results of the CCA corroborate the data obtained by the Pearson correlation tests for the relation between the environmental variables and the species. The response of beetles and bugs to the increase in conductivity seemed to define, at least, the species typical of sites with high values of this variable. Some species respond negatively to high conductivity (Garrido & Munilla, 2008), while others are adapted to sudden changes in this variable and can be considered tolerant. In this sense, ordination analysis identified a group of tolerant species (i. e. *Berosus hispanicus*, *Rhantus suturalis* and *Sigara stagnalis*), typical of habitats with high conductivity or salinity values (Arnold & Ormerod, 1997; Garrido & Munilla, 2008;

Hansen, 1987; Martinoy et al., 2006). Different authors have identified salinity as the most important factor regulating species composition (Britton & Johnson, 1987; Cognetti & Maltagliati, 2000; Martinoy et al., 2006; Timms, 1993). Thus, the two samples with higher salinities (Vixán in spring and summer of 1998) appear separated from the rest of the samples of 1998 (Figure 7).

In this study, *Gyrinus caspius* was positively correlated with temperature. In the literature there are no references concerning the correlation of this species with this factor. However, it is mainly found near the sea (Hansen, 1987), in lentic freshwater or subsalt systems (Sánchez-Fernández et al., 2007), and in high salt marshes (Bigot & Marazanof, 1966; Montes et al., 1982).

Some species, such as *Helochares puntactus* and *Hygrobia hermanni*, were not found in the oligohaline water body (Vixán). Both species were present only in the fresh water systems (Bodeira and Xuño), which could suggest their presence in waters with less conductivity. *H. puntactus* inhabits margins of ponds and lakes, generally eutrophic with grasses and other submerged macrophytes (Valladares, 1995) and pools in heathlands (Aquilina, 2010). On the other hand, Bigot & Marazanof (1966) and Montes et al. (1982) found *H. hermanni* in eutrophic and high salinity waters in the Guadalquivir Marshes (South of Spain).

The Bray-Curtis coefficient revealed significant differences in terms of faunal affinity between the two periods of time. The three lagoons studied presented significantly different assemblages 10 years later. The response of Coleoptera and Hemiptera assemblages to the passing of the time (10 years later) was a change in species composition, with the substitution of some species by others, not by changes in the richness values, as these were similar between the two periods of time. The change in species composition found in these ecosystems seems to be a consequence of the high temporal variability. The heterogeneity in physiography and hydrology of the coastal lagoons influences the faunal composition, and variations in the physicochemical characteristics can change the composition of the aquatic assemblages over time. These special conditions of the coastal lagoons were also mentioned by Martinoy et al. (2006)

In this sense, it is important to note that certain species occurred in only one of the years. Species such as *Enochrus halophilus* and *Sigara stagnalis* were very common in 1998 and were absent in 2008. Others, including *Hydrovatus clypealis*, *Paracymus scutellaris*, *Enochrus fuscipennis* and *Anisops sardeus sardeus*, were abundant in the three lagoons in 2008, but not found in 1998. *A. s. sardeus* is a trans-Iberian hemipteran, present in several environments, including moderately mineralised waters, and in artificial or newly created water bodies (Abellán et al., 2004), very common in the south of the Iberian Peninsula (Nieser & Montes, 1984).

In the same genus, certain species were substituted by another 10 years later. For example, in the genus *Graptodytes* three species were found in 1998 (*G. aequalis*, *G. flavipes*, *G. ignotus*), but only *G. varius* was collected in 2008. The same occurs with *Dryops*, which changes *D. striatellus* (1998) for *D. luridus* (2008), and *Enochrus*, in which *E. fuscipennis* substituted *E. halophilus*. In the Iberian Peninsula, *D. luridus* is much more common than *D. striatellus* (Montes & Soler, 1986). In fact, in England *D. striatellus* is a vulnerable species (Bilton et al., 2009). According to Hansen (1987) both *E. halophilus* and *E. fuscipennis* are found in temporary brackish pools above the high tide line or occasionally also in slightly saline water. *E. halophilus* is classified as "vulnerable" on the red list of Irish threatened species

(IUCN) (Foster et al., 2009). This species was found in wetlands in the Mediterranean coast of Spain in eutrophic or even in high salinity waters (Bigot & Marazanof, 1966; Ribera et al. 1996). *E. fuscipennis* has been found in stagnant water rich in organic matter (Valladares, 1995). According to Bilton et al. (2009) this species is widespread in ponds and other wetlands throughout the UK and north-west Europe.

The aquatic bug *Plea minutissima* is one of the most abundant species in this study. Other studies have also reported a high number of individuals of this insect in stagnant water bodies (Bloechl et al., 2010; González & Valladares, 1996). Another very abundant and common species of Hemiptera was *Hesperocorixa linnaei* (1,360 specimens recorded in 75% of the samples), which is different to the data obtained by Martinoy et al. (2006) in a study conducted in 84 water bodies in coastal ecosystems of the Empordà wetlands (NE Spain), where this species presented a very low abundance.

Some of the species found in these lagoons are rare and not common in other regions. For example, the case of *Ochthebius viridis fallaciosus*, which was the most abundant species of Coleoptera (1,961 individuals, recorded in 75% of the samples) should be noted. According to Foster et al. (2009), this subspecies is on the red list of threatened species (IUCN) in Ireland, classified as "near threatened", i.e. a species does not qualify for Critically Endangered, Endangered or Vulnerable now, but is close to qualifying for or is likely to qualify for a threatened category in the near future. In this country it is a sparsely distributed species that appears to be confined to brackish water.

This study demonstrates the importance of the conservation of these lagoons for maintaining aquatic biodiversity. In this regard, it should be said that the Natura 2000 network is effective in protecting wildlife, as far as these lagoons are concerned. If we compare data from 1998 to those obtained 10 years later, we note that there were no significant changes in the values of the diversity indices. In terms of faunal composition, it can be said that the observed changes are mainly due to instability, typical of the coastal lagoons. This can be highlighted by the specific changes in certain variables, such as the case of increased salinity in the summer of 1998.

Coleoptera and Hemiptera assemblages are good indicators of the environmental quality of coastal lagoons. Their responses to changes in environmental conditions differ. Some species are adapted to increasing in certain variables, for example, *Berosus hispanicus* and *Rhantus suturalis* correlated with high salinity. To conclude, it is important to note that the preservation of species is one of the main factors that should be taken into consideration in developing monitoring systems and assessing reference conditions, as stipulated by the Water Framework Directive (Council of the European Communities 2000).

7. References

Abbiati, M. & Basset, A. (2001). Ecological research and conservation of coastal ecosystems. *Aquatic Conservation: Marine and Freshwater Ecosystems*, 11, 233-234.

Abellán, P.; Sánchez-Fernández, D.; Barahona, J.; Velasco, J. & Millán, A. (2004). Coleópteros y hemípteros acuáticos con interés de conservación en el entorno de Calblanque. *Actas III Congreso de la Naturaleza de la Región de Murcia*, 89-97.

Aliaume, C.; Do Chi, T.; Viaroli, P. & Zaldívar, J.M. (2007). Coastal lagoons of Southern Europe: recent changes and future scenarios. *Transitional Waters Monographs*, 1, 1-12.

Álvarez-Cobelas, M.; Catalán, J.; García de Jalón, D.; Alonso, M.; Amores, V.; Armengol, J.; Artigas, J.; Barquín, J.; Bayo, M.; Boix, D.; Butturini, A.; Camacho, A.; Carrillo, P.; Casas, J.; Cirujano, S.; Comín, F.; Cruz-Pizarro, L.; Gaudes, A.; Lucena, J.; Menéndez, M.; Morales, R.; Moreno-Ostos, E.; Muñoz, I.; Pérez, C.; Pozo, J.; Quintana, X.; Reche, I.; Rodrigo, M.; Romaní, A.; Sabater, S.; Sánchez-Castillo, P.; Soria, J.M.; Suárez, M.L.; Bernal, S.; Sabater, F.; Toja, J.; Vega, J.C.; Vicente, I. & Vidal Abarca, M.R. (2005). Impactos sobre los ecosistemas acuáticos continentales, In: *Impactos del cambio climático en España*, pp. 113-146, Ministerio de Medio Ambiente, Madrid.

Aquilina, R. (2010). An assemblage of Coleoptera from ponds on Povington Heath, Dorset. *The Coleopterist,* 19(1), 25-29.

Arnold, S.L. & Ormerod, S.J. (1997). Aquatic macroinvertebrates and environmental gradients in Phragmites reedswamps: implications for conservation. *Aquatic Conservation: Marine and Freshwater Ecosystems*, 7, 153-163.

Arocena, R. (2007). Effects of Submerged Aquatic Vegetation on Macrozoobenthos in a Coastal Lagoon of the Southwestern Atlantic. *International Review of Hydrobiology*, 92(1), 33-47.

Barnes, R.S.K. (1999). The conservation of brackish-water systems: priorities for the 21st century. *Aquatic Conservation: Marine and Freshwater Ecosystems*, 9, 523-527.

Basset, A. (2007). Ecosystems and Society: do they really need to be bridged? *Aquatic Conservation: Marine and Freshwater Ecosystems*, 17, 551-553.

Basset, A. & Abbiati, M. (2004). Challenges to transitional water monitoring: ecological descriptors and scales. *Aquatic Conservation: Marine and Freshwater Ecosystems*, 14, S1-S3.

Basset, A.; Sabetta, L. & Carrada, G.C. (2006a). Conservation of transitional water ecosystems in the Mediterranean area: bridging basic ecological research and theories with requirements of application. *Aquatic Conservation: Marine and Freshwater Ecosystems*, 16, 439-440.

Basset, A.; Sabetta, L.; Fonnesu, A.; Mouillot, D.; Do Chi, T.; Viaroli, P.; Giordano, G.; Reizopoulou, S.; Abbiati, M. & Carrada, G.C. (2006b). Typology in Mediterranean transitional waters: new challenges and perspectives. *Aquatic Conservation: Marine and Freshwater Ecosystems*, 16, 441-455.

Bigot, L. & Marazanof, F. (1966). Notes sur l'ecologie des Coléoptères aquatiques des Marismas du Guadalquivir et premier inventaire des Coléoptères et Lépidoptères du Coto Doñana (Andalucía). *Annales de Limnologie,* 2, 491-502.

Bilton, D.T.; McAbendroth, L.; Bedford, A. & Ramsay, P.M. (2006). How wide to cast the net? Cross-taxon congruence of species richness, community similarity and indicator taxa in ponds. *Freshwater Biology*, 51, 578–590.

Bilton, D.T.; McAbendroth, L.C.; Nicolet, P.; Bedford, A.; Rundle, S.R. ; Foggo, A. & Ramsay, P.M. (2009). Ecology and conservation status of temporary and fluctuating ponds in two areas of southern England. *Aquatic Conservation: Marine and Freshwater Ecosystems*, 19, 134–146.

Bloechl, A.; Koenemann, S.; Philippi, B. & Melber, A. (2010). Abundance, diversity and succession of aquatic Coleoptera and Heteroptera in a cluster of artificial ponds in the North German Lowlands. *Limnologica*, 40, 215-225.

Boix, D.; Gascón, S.; Sala, J.; Badosa, A. Brucet, S. López-Flores, R.; Martinoy, M.; Gifre, J. & Quintana, X. D. (2008). Patterns of composition and species richness of crustaceans and aquatic insects along environmental gradients in Mediterranean water bodies. *Hydrobiologia* 597, 53–69.

Britton, R.H. & Johnson, A.R. (1987). An Ecological Account of a Mediterranean Salina: The Salin de Giraud, Camargue (S. France). *Biological Conservation*, 42, 185-230.

Broering, U. & Niedringhaus, R. (1988). On the ecology of aquatic Hemiptera (Hemiptera: Nepomorpha) in small ponds on the East Frisian Island of Norderney (FRG). *Fundamental and applied Limnology, Archiv für Hydrobiologie*, 111, 559-574.

Casado, S. & Montes, C. (1995). *Guía de los lagos y humedales de España*. J.M. Reyero, Madrid.

Cognetti, G. & Maltagliati, F. (2000). Biodiversity and Adaptative Mechanisms in Brackish Water Fauna. *Marine Pollution Bulletin*, 40, 7-14.

Davis, J.A.; Rolls, S.W. & Balla, S.A. (1987). The role of the Odonata and aquatic Coleoptera as indicators of environmental quality in wetlands, In: *The Role of Invertebrates in Conservation and Biological Survey*, J.D. Majer (ed.), pp. 31-42, Western Australian Department of Conservation and Land Management, Perth.

Della Bella, V.; Bazzanti, M.; Dowgiallo, M.G. & Iberite, M. (2008). Macrophyte diversity and physico-chemical characteristics of Tyrrhenian coast ponds in central Italy: implications for conservation. *Hydrobiologia*, 597, 85–95.

Diehl, S. & Kornijów, R. (1998). Influence of submerged macrophytes on trophic interactions among fish and macroinvertebrates, In: *Structuring Role of Submerged Macrophytes in Lakes*, E. Jeppesen, M. Søndergaard, M. Søndergaard & K. Christoffersen (eds.), pp. 24-46, Springer-Verlag, New York.

Drake, D.C.; Kelly, D. & Schallenberg, M. (2011). Shallow coastal lakes in New Zealand: current conditions, catchment-scale human disturbance, and determination of ecological integrity. *Hydrobiologia*, 658, 87-101.

Duigan, C.A.; Allot, T.E.H.; Monteith, D.T.; Patrick, S.T.; Lancaster, J. & Seda, J.M. (1998). The ecology and conservation of Llyn Idwal and Llyn Cwellyn (Snowdonia National Park, North Wales, U.K.)- two lakes proposed as Special Areas of Conservation in Europe. *Aquatic Conservation: Marine and Freshwater Ecosystems*, 8, 325-360.

Esteves, F.A.; Caliman, A.; Santangelo, J.M.; Guariento, R.D.; Farjalla, V.F. & Bozelli, R.L. (2008). Neotropical coastal lagoons: An appraisal of their biodiversity, functioning, threats and conservation management. *Brazilian Journal of Biology*, 68(4), 967-981.

Eyre, M.D. & Foster, G.N. (1989). A comparison of aquatic Hemiptera and Coleoptera communities as a basis for environmental and conservation assessments in static water sites. *Journal of Applied Entomology*, 108, 355-362.

Eyre, M.D. & Rushton, S.P. (1989). Quantification of conservation criteria using invertebrates. *Journal of Applied Entomology*, 26, 159-171.

Eyre, M.D.; Foster, G.N. & Young, A.G. (1993). Relationships between water-beetle distributions and climate variables: a possible index for monitoring global climatic change. *Fundamental and applied Limnology, Archiv für Hydrobiologie*, 127, 437-450.

Florencio, M.; Serrano, L.; Gómez-Rodríguez, C.; Millán, A. & Díaz-Paniagua, C. (2009). Inter- and intra-annual variations of macroinvertebrate assemblages are related to the hydroperiod in Mediterranean temporary ponds. *Hydrobiologia*, 634, 167–183.

Foster, G.N. (1987). The use of Coleoptera records in assessing the conservation status of wetlands, In: *The Use of Invertebrates in the Site Assessment for Conservation*, M.L. Luff (ed.), pp. 8-18, University of Newcastle upon Tyne, UK.

Foster, G.N. (1999). Biodiversity action plans for British water beetles. *Latissimus*, 11, 1-13.

Foster, G.N. (2000). The aquatic Coleoptera of British saltmarshes: extremes of generalism and specialism, In: British Saltmarshes, B.R. Sherwood, B.G. Gardiner & T. Harris (eds.), pp. 223-233, Linnean Society, London.

Foster, G.N.; Foster, A.P.; Eyre, M.D. & Bilton, D.T. (1990).Classification of water beetles assemblages in arable fenland and ranking of sites in relation to conservation value. *Freshwater Biology*, 22, 343-354.

Foster, G.N.; Nelson, B.H. & O'Connor, A. (2009). *Ireland Red List No. 1 – Water beetles.* National Parks and Wildlife Service, Department of Environment, Heritage and Local Government, Dublin, Ireland.

Garrido, J.; Sáinz-Cantero, C.E. & Díaz, J.A. (1996). Fauna entomológica del Parque Nacional de Doñana (Huelva, España) I. (Coleoptera, Polyphaga). *Nouvelle Revue d' Entomologie*, 13, 57–71.

Garrido, J.; Sáinz-Cantero, C.E. & Régil-Cueto, J.A. (1997). Fauna entomológica del Parque Nacional de Doñana (Huelva, España) II. (Coleoptera, Adephaga). *Nouvelle Revue d' Entomologie*, 14(4), 365–377.

Garrido, J. & Munilla, I. (2008). Aquatic Coleoptera and Hemiptera assemblages in three coastal lagoons of the NW Iberian Peninsula: assessment of conservation value and response to environmental factors. *Aquatic Conservation: Marine and Freshwater Ecosystems*, 18, 557-569.

Gaston, K.J. (1994). *Rarity.* Chapman & Hall, London.

González, S.C. & Valladares, L.F. (1996). The community of Odonata and aquatic Heteroptera (Gerromorpha and Nepomorpha) in a rehabilitated wetland: the Laguna de la Nava (Palencia, Spain). *Fundamental and applied Limnology, Archiv für Hydrobiologie*, 136(1), 89-104.

Goudie, A. (1990). *The Human Impact on the Natural Environment.* Basil Blackwell, Oxford.

Greenwood, M.T. & Wood, P.J. (2002). Effects of seasonal variation in salinity on a population of *Enochrus bicolor* Fabricius, 1792 (Coleoptera: Hydrophilidae) and implications for other beetles of conservation interest. *Aquatic Conservation: Marine and Freshwater Ecosystems*, 13, 21-34.

Hansen, M. (1987). *The Hydrophiloidea (Coleoptera) of Fennoscandia and Denmark. Fauna Entomologica Escandinavica, Vol. 18,* Scandinavian Science Press, ISBN 87-87491-35-4, Copenhagen, Denmark

Heino, J. (2000). Lentic macroinvertebrate assemblage structure along gradients in spatial heterogeneity, habitat size and water chemistry. *Hydrobiologia*, 418, 229–242.

Hufnagel, L.; Bakonyi, G. & Vásárhelyi, T. (1999). New approach for habitat characterization based on species lists of aquatic and semiaquatic bugs. *Environmental Monitoring and Assessment*, 58, 305–316.

IPCC (2007). *Climate Change 2007: Synthesis Report. Contribution of Working Groups I, II and III to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change*, Core Writing Team, R.K. Pachauri & A. Reisinger (eds.), IPCC, Geneva, Switzerland.

Jansson, A. (1977). Micronectae (Heteroptera, Corixidae) as indicators of water quality in two lakes in southern Finland. *Annales Zoologici Fennici*, 14, 118–124.

Jones, T.A. & Hughes, J.M.R. (1993). Wetland inventories and wetlands loss studies: a European perspective, In: *Waterfowl and Wetland Conservation in the 1990s: a Global Perspective*, M. Moser, R.C. Prentice & J. Van Vessem (eds.), pp. 164-169, IWRB Special Publication, Slimbridge, UK.

Kirk, R.M. & Lauder, G.A. (2000). *Significant coastal lagoon systems in the South Island, New Zealand: coastal processes and lagoon mouth closure*. N.Z. Department of Conservation, Wellington.

Kjerfve, B. (1994). Coastal Lagoons, In: *Coastal Lagoon Processes*, B. Kjerfve (ed.), pp. 1-8, Elsevier Science Publishers, Amsterdam.

Kjerfve, B. & Magill, K.E. (1989). Geographic and hydrodynamic characteristics of shallow coastal lagoons. *Marine Geology*, 88, 187-199.

Lancaster, J. & Scudder, G.G.E. (1987). Aquatic Coeloptera and Hemiptera in some Canadian saline lakes: patterns in community structure. *Canadian Journal of Zoology*, 65, 1383–1390.

Langenheder, S.; Kisand, V.; Wikner, J. & Tranvik, L.J. (2003). Salinity as a structuring factor for the composition and performance of bacterioplankton degrading riverine DOC. *FEMS Microbiology Ecology,* 45, 189-202.

Lewis, W.M. Jr. (1983). A revised classification of lakes based on mixing. *Canadian Journal of Fisheries and Aquatic Sciences,* 40, 1779-1787.

Lucena-Moya, P.; Abraín, R.; Pardo, I.; Hermida, B. & Domínguez, M. (2010). Invertebrate species list of coastal lagoons in the Balearic Islands. *Transitional Waters Bulletin*, 4(1), 1-11.

Maezono, Y. & Miyashita, T. (2003). Community level impacts induced by introduced largemouth bass and bluegill in farm ponds in Japan. *Biological Conservation*, 109(1), 111–121.

Margules, C.R.; Nicholls, A.O. & Pressey, R.L. (1988). Selecting networks of reserves to maximise biological diversity. *Biological Conservation*, 43, 63–76.

Martinoy, M.; Boix, D.; Sala, J.; Gascón, S.; Gifre, J.; Argerisch, A.; de la Barrera, R.; Brucet, S.; Badosa, A.; López-Flores, R.; Méndez, M.; Utgé, J.M. & Quintana, X.D. (2006). Crustacean and aquatic insect assemblages in the Mediterranean coastal ecosystems of Empordà wetlands (NE Iberian Peninsula). *Limnetica*, 25, 665-682.

McCabe, D.J. & Gotelli, N.J. (2000). Effects of disturbance frequency, intensity, and area on assemblages of stream macroinvertebrates. *Oecologia*, 124, 270-279.

Millán, A., Moreno, J.L. & Velasco, J. (1997). Coleópteros y Heterópteros acuáticos del complejo lagunar del río Arquillo (Albacete). *Separatas de al-basit. Revista de estudios albacetenses*, 40.

Millán, A.; Moreno, J.L. & Velasco, J. (2001). Estudio faunístico y ecológico de los coleópteros y heterópteros acuáticos de las lagunas de Albacete. *Separatas de Sabuco. Revista de estudios albacetenses*, 1, 43–94.

Millán, A.; Moreno, J.L. & Velasco, J. (2002). *Los Coleópteros y Heterópteros acuáticos y semiacuáticos de la provincia de Albacete. Catálogo faunístico y estudio ecológico*, Instituto

de Estudios Albacetenses Don Juan Manuel de la Excma. Diputación Provincial de Albacete, Albacete.

Mitsch, W.J. & Gosselink, J.G. (2000). *Wetlands*, John Wiley and Sons, New York.

Montes, C. ; Ramirez, L. & Soler, A. (1982). Variación estacional de las taxocenosis de odonatos, coleópteros y heterópteros acuáticos en algunos ecosistemas del Bajo Guadalquivir (SW. España) durante un ciclo anual. *Anales de la Universidad de Murcia, Ciencias*, 38(1-4), 19-100.

Montes, C. & Soler, A.G. (1986). *Lista faunística y bibliográfica de los coleópteros acuáticos Dryopoidea (Dryopidae & Elmidae) de la Península Ibérica e Islas Baleares. Listas de la flora y fauna de las aguas continentales de la Península Ibérica*. Asociación Española de Limnología, Madrid.

Moreno, J.L.;Millán, A.; Suárez, M.L.; Vidal-Abarca, M.R. & Velasco, J. (1997). Aquatic Coleoptera and Hemiptera assemblages in water bodies from ephemeral coastal streams ('ramblas') of south-eastern Spain. *Fundamental and applied Limnology, Archiv für Hydrobiologie*, 141, 93–107.

Nichols, M.M. (1989). Sediment accumulation rates and relative sea-level rise in lagoons. *Marine Geology*, 88, 201-219.

Nichols, M. & Allen, G. (1981). Sedimentary processes in coastal lagoons. *UNESCO Technical Bulletin in Marine Science*, 33, 27-80.

Nieser, N. & Montes, C. (1984). *Lista faunística y bibliográfica de los Heterópteros acuáticos (Nepomorpha & Gerromorpha) de España y Portugal*. Asociación Española de Limnología, Listas de la flora y fauna de las aguas continentales de la Península Ibérica. Madrid.

Nieser, N.; Baena, M.; Martínez-Avilés, J. & Millán, A. (1994). *Clave para la identificación de los heterópteros acuáticos (Nepomorpha & Gerromorpha) de la Península Ibérica – Con notas sobre las especies de las islas Azores, Baleares, Canarias y Madeira. Claves de identificación de la Flora y Fauna de las aguas continentales de la Península Ibérica*. Asociación Española de Limnología, Madrid.

Nixon, S.W. (1995). Coastal eutrophication: a definition, social causes, and future concerns. *Ophelia*, 41, 199–220.

Oertli, B.; Joye, D.A.; Castella, E.; Juge, R.; Lehmann, A. & Lachavanne, J-B. (2005). PLOCH: a standardized method for sampling and assessing the biodiversity in ponds. *Aquatic Conservation: Marine and Freshwater Ecosystems*, 15, 665-679.

Orfanidis, S.; Reizopoulou, S. & Basset, A. (2008). Transitional states in transitional waters. *Aquatic Conservation: Marine and Freshwater Ecosystems*, 18, S1-S3.

Pérez-Bilbao, A. & Garrido, J. (2009). Evaluación del estado de conservación de una zona LIC (Gándaras de Budiño, Red Natura 2000) usando los coleópteros acuáticos como indicadores. *Limnetica*, 28(1), 11-22.

Pérez-Bilbao, A.; Benetti, C.J. & Garrido, J. (2009). Nuevas aportaciones al conocimiento de la familia Hydrochidae (Coleoptera) en Galicia (Noroeste de España). *Boletín de la Sociedad Entomológica Aragonesa*, 45, 241-244.

Pérez-Bilbao, A.; Benetti, C.J. & Garrido, J. (2010). Coleópteros acuáticos (Polyphaga, Hydrophilidae) en lagunas de la Red Natura 2000 de Galicia (Noroestes de España). *Boletín de la Sociedad Entomológica Aragonesa*, 46, 255-260.

Piccini, C. & Conde, D. (2004). Cambios drásticos en la comunidad bacteriana de la laguna de Rocha y sus posibles implicaciones ambientales. *Revista Agrociencia*, 9(1-2), 269-275.

Polhemus, J.T. & Polhemus, D.A. (2008). Global diversity of true bugs (Heteroptera; Insecta) in freshwater. *Hydrobiologia*, 595, 379-391.

Ponti, M.; Casselli, C. & Abbiati, M. (2011). Anthropogenic disturbance and spatial heterogeneity of macrobenthic invertebrate assemblages in coastal lagoons: the study case of Pialassa Baiona (northern Adriatic Sea). *Helgoland Marine Research*, 65, 25-42.

Ramil, P.; Cillero, C.; García, N.; Codesido, D.; Rubinos, M. & Ferreiro, J. (2007). *Evaluación del estado de conservación de las lagunas costeras de Galicia: Propuestas de conservación y uso sostenible*. Instituto de Biodiversidade Agraria e Desenvolvemento Rural (IBADER), Lugo, Spain.

Ribera, I. & Foster, G.N. (1992). Uso de coleópteros acuáticos como indicadores biológicos (Coleoptera). *Elytron*, 6, 61-75.

Ribera, I.; Bilton, D.T.; Aguilera, P. & Foster, G.N. (1996). A north African-European transition fauna: water beetles (Coleoptera) from the Ebro Delta and other Mediterranean coastal wetlands in the Iberian Peninsula. *Aquatic Conservation: Marine and Freshwater Ecosystems*, 6, 121-140.

Ribera, I.; Hernando, C. & Aguilera, P. (1999). An annotated checklist of the Iberian water beetles (Coleoptera). *Zapateri, Revista aragonesa de entomología*, 8, 43-111.

Ribera, I. & Vogler, A.P. (2000). Habitat type as a determinant of species range sizes: the example of lotic–lentic differences in aquatic Coleoptera. *Biological Journal of the Linnean Society*, 71, 33-52.

Ribera, I. & Vogler, A.P. (2004). Speciation of Iberian diving beetles in Pleistocene refugia (Coleoptera, Dytiscidae). *Molecular Ecology,* 13, 179–193.

Ricciardi, A. (2006). Are modern biological invasions an unprecedent form of global change? *Conservation Biology*, 21(2), 329–336.

Ricciardi, A.; Whoriskey, F.G. & Rassmussen, J.B. (1997). The role of the zebra mussel (Dreissena polymorpha) in structuring macroinvertebrate communities on hard substrata. *Canadian Journal of Fisheries and Aquatic Sciences*, 54(11), 2596–2608.

Roback, S.S. (1974). Insects (Arthropoda: Insecta), In: *Pollution Ecology of Freshwater Invertebrates*, C.W. Hart Jr., & S.L.H. Fuller (eds.), pp. 313-376, Academic Press, Inc., New York.

Rodríguez-Pérez, H.; Florencio, M.; Gómez-Rodríguez, C.; Green, A.J.; Díaz-Paniagua, C. & Serrano, L. (2009). Monitoring the invasion of the aquatic bug *Trichocorixa verticalis verticalis* (Hemiptera: Corixidae) in the wetlands of Doñana National Park (SW Spain). *Hydrobiologia*, 634, 209-217.

Sáinz-Cantero, C.E. & Garrido, J. (1996). Primera cita de *Cymbiodyta marginella* (Fabricius, 1792) en España (Col., Hydrophilidae). *Bulletin de la Societé entomologique de France*, 101, 508.

Sánchez-Fernández, D.; Abellán, P.; Mellado, A.; Velasco, J. & Millán, A. (2006). Are water beetles good indicators of biodiversity in Mediterranean aquatic ecosystems? The case of the Segura river basin (SE Spain). *Biodiversity and Conservation*, 15, 4507-4520.

Sánchez-Fernández, D.; Abellán, P.; Camarero, F.; Esteban, I.; Gutiérrez-Cánovas, C.; Ribera, I.; Velasco, J. & Millán, A. (2007). Los macroinvertebrados acuáticos de las salinas de Añana (Álava, España): biodiversidad, vulnerabilidad y especies indicadoras. *Boletín de la Sociedad Entomológica Aragonesa*, 40, 233-245.

Savage, A.A. (1996). Density dependent and density independent relationships during a twenty-seven year study of the population dynamics of the benthic macroinvertebrate community of a chemically unstable lake. *Hydrobiologia*, 335, 115-131.

Shine, C. & Klemm, C. (1999). *Wetlands, Water and the Law: Using law to Advance Wetland Conservation and Wise Use*, IUCN, Gland.

Solimini, A.G.; Bazzanti, M.; Ruggiero, A. & Carchini, G. (2008). Developing a multimetric index of ecological integrity based on macroinvertebrates of mountain ponds in central Italy. *Hydrobiologia*, 597, 109-123.

Søndergaard, M. & Moss, B. (1998). Impact of submerged macrophyteson phytoplankton in shallow freshwater lakes, In: *Structuring Role of Submerged Macrophytes in Lakes*, E. Jeppesen, M. Søndergaard, M. Søndergaard & K. Christoffersen (eds), pp. 115-133, Springer-Verlag, New York.

Soria, J.M. & Sahuquillo, M. (2009). 1150 Lagunas costeras (*), In: *Bases ecológicas preliminares para la conservación de los tipos de hábitats de interés comunitario en España*. Ministerio de Medio Ambiente y Medio Rural y Marino, Madrid, España.

Stenert, C. & Maltchik, L. (2007). Influence of area, altitude and hydroperiod on macroinvertebrate communities in southern Brazil wetlands. *Marine and Freshwater Research*, 58, 993-1001.

Sylaios, G. & Theocharis, V. (2002). Hydrology and Nutrient Enrichment at Two Coastal Lagoon Systems in Northern Greece. *Water Resources Management*, 16, 171-196.

Ter Braak, C.J.F. & Šmilauer, P. (2002). *CANOCO Reference manual and CanoDraw for Window's User's guide: Software for Canonical Community Ordination (version 4.5)*. Microcomputer Power, Ithaca.

Timms, B.V. (1993). Saline lakes of the Paroo, inland New South Wales, Australia. *Hydrobiologia*, 267, 269-289.

Tuly, O.; McCarthy, T.K. & O'Donell, D. (1991). The ecology of the Corixidae (Hemiptera: Hemiptera) in the Corrib catchment, Ireland. *Hydrobiologia*, 210, 161-169.

Valladares, L.F. (1995). Los Palpicornia acuáticos de la provincia de León III. Helophoridae, Hydrochidae e Hydrophilidae (Coleoptera). *Boletín de la Asociación española de Entomología*, 19(1-2), 281-308.

Vierssen, W. & Verhoeven, J.T.A. (1983). Plant and animal communities in brackish supralitoral pools ('dobben') in the northern part of the Netherlands. *Hydrobiologia*, 98, 203-211.

Ward, L.G. & Ashley, G.M. (1989). Introduction: coastal lagoonal systems. *Marine Geology*, 88, 181-185.

Williams, P.; Whitfield, M.; Biggs, J.; Bray, S.; Fox, G.; Nicolet, P. & Sear, D. (2003). Comparative biodiversity of rivers, streams, ditches and ponds in an agricultural landscape in Southern England. *Biological Conservation*, 115, 329-341.

Witte, F.; Msuku, B.S.; Wanink, J.H.; Seehausen, O.; Katunzi, E.F.B.; Goudswaard, P.C. & Goldsmidt, T. (2000). Recovery of cichild species in lake Victoria: an examination of factors leading differential extinction. *Reviews in Fish Biology and Fisheries,* 10, 233–241.

Zaldívar, J.M.; Cardoso, A.C.; Viaroli, P.; Newton, A.; de Wit, R.; Ibáñez, C.; Reizopoulou, S.; Somma, F.; Razinkovas, A.; Basset, A.; Holmer, M. & Murray, N. (2008). Eutrophication in transitional waters: an overview. *Transitional Waters Monographs*, 1, 1-78.