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ARTICLE

Farm Ponds as Potential Complementary Habitats to Natural Wetlands in a Mediterranean Region

J. Jesús Casas · Julia Toja · Patricio Peñalver · Melchor Juan · David León · Francisca Fuentes-Rodríguez · Irene Gallego · Encarnación Fenoy · Carmen Pérez-Martínez · Pedro Sánchez · Santiago Bonachela · M. Antonia Elorrieta

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Abstract We compared morphometric and physico-chemical characteristics of farm ponds and natural wetlands in Andalusia (southern Spain) to determine whether artificial waterbodies might act as alternative and/or complementary habitats for aquatic biodiversity. Farm ponds were much smaller than natural wetlands, making them unsuitable for species requiring large waterbodies. However, we observed high farm pond density in areas lacking natural wetlands, which suggests a prime role for the conservation of species with low dispersal capacities. Natural-substrate ponds were abundant in traditional extensive farming systems and showed shoreline complexity as high as the most complex natural wetlands. Areas with more intensive agriculture were dominated by artificial-substrate ponds and wetlands, with low physical complexity in both. The high copper load in sediments, due to the use of copper sulphate as biocide, differentiated the artificial-substrate ponds from natural-looking ponds and all natural wetland types. Aqueous mineral levels in farm ponds were much lower than in natural wetlands. We can conclude that farm ponds might play a principal role in region-wide habitat complementarity, by providing a relatively high density of small, permanent, oligohaline waterbodies that is not matched by natural wetland. To enhance this role, measures regulating both pond construction and management are needed, particularly for artificial-substrate ponds.

Keywords Biodiversity conservation · Constructed wetlands · Habitat quality · Wetland creation

J. J. Casas (✉) · M. Juan · F. Fuentes-Rodríguez · I. Gallego · E. Fenoy
Departamento de Biología Vegetal y Ecología, Universidad de Almería, 04120 Almería, Spain
e-mail: jjcasas@ual.es

J. Toja · P. Peñalver · D. León
Departamento de Biología Vegetal y Ecología, Universidad de Sevilla, Seville, Spain

C. Pérez-Martínez
Instituto del Agua, Universidad de Granada, Granada, Spain

P. Sánchez
Departamento de Botánica, Universidad de Granada, Granada, Spain

S. Bonachela
Departamento de Producción Vegetal, Universidad de Almería, 04120 Almería, Spain

M. A. Elorrieta
Departamento de Fitopatología, Labcolor-COEXPHAL, Almería, Spain

Introduction

Agricultural intensification over the past century has caused widespread declines of biodiversity in traditionally managed farmlands (Benton et al. 2003). In Spain, increasing agricultural intensification has taken place during recent decades particularly in Mediterranean areas with mild winters, prompted by new irrigation technologies and policy and funding changes. However, in spite of this general intensification, areas with traditional farmland systems, highly valuable for biodiversity conservation, still persist in Spain.

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Perhaps one of the most detrimental environmental impacts of agricultural expansion and intensification has been the loss of natural wetlands (OECD 1996). For Spain, Casado and Montes (1995) indicate that more than 60% of wetlands disappeared in the last century. In Andalusia (southern Spain), Reques (2006) estimates that 45% and 61% of large and small wetlands, respectively, had been lost over the previous five decades. Agricultural intensification, urbanization, aquifer overexploitation, or drainage programs to eradicate malaria are among the main current or historical factors responsible for wetland loss and degradation in southern Spain (Serrano and Serrano 1996; Sousa et al. 2009; Casas et al. 2011a).

As in other semiarid regions (e.g., Lutton et al. 2010), the expansion of irrigated agriculture in Andalusia has promoted the proliferation of small in-farm ponds to secure access to water and overcome the problem of an irregular natural water supply (Casas et al. 2011b). The value of small artificial waterbodies for the conservation of aquatic biodiversity in agricultural landscapes has already been recognized in different climatic regions. These provide alternative habitats for many species that might otherwise have disappeared due to the accelerated loss and degradation of their natural wetland habitats (Hazell et al. 2004; Williams et al. 2004; Céréghino et al. 2008). However, several studies have pointed out that the morphological, physical, and chemical habitat characteristics of the new in-farm ponds may make them unsuitable as a replacement for natural wetlands (Sánchez-Zapata et al. 2005; Brainwood and Burgin 2006; Markwell and Fellows 2008).

This study aims to evaluate and compare structural, hydrological, and chemical characteristics, potentially important for biodiversity conservation, of farm ponds and natural wetlands in Andalusia (Southern Spain). This comparison is mainly intended to establish whether farm ponds might provide alternative and/or complementary aquatic habitats for biodiversity conservation in the varied agricultural landscapes of this region.

Study Area

Our study was carried out in Andalusia (Spain), a region of 87,597 km^2 located in the southernmost part of continental Europe (between parallels 36ºN and 38º44′ N) (Fig. 1). This is a geologically diverse region that can be roughly divided into two main areas separated by the Guadalquivir River axis: the western and northern parts dominated by acidic rocks of the Sierra Morena mountain ranges, and the south eastern part by calcareous-dolomitic rocks of the Betic ranges. In this second area, several high mountains (over 2000 m a.s.l.) are crowned by schist, and valleys with floors covered by recent marine deposits (marls) are frequent. The climate is Mediterranean, with precipitation regimes varying from sub-humid in the west and mountain ranges to semiarid in many eastern lowland areas (Table 1).

Around 80% of human water use and nearly 10,000 km^2 are dedicated to irrigated agriculture in Andalusia, which combined with the general aridity of the region, has determined the proliferation of many irrigation facilities, including thousands of in-farm ponds (Casas et al. 2011b) (Fig. 1).

Andalusia harbors around 17% of the total number of natural wetlands in Spain, rising to 56% when the total wetland surface area is considered (Consejería de Medio Ambiente 2005). This is a highly diverse set of wetlands which covers from high mountain tarns over 3000 m a.s.l. in the Sierra Nevada mountains (Betic mountain Range, southeast of Andalusia) to coastal lagoons and marshes, including the largest marshes in Spain (Doñana) of high international importance for the conservation of migrant water birds (Fig. 1) (Consejería de Medio Ambiente 2005).

Methods

Data Collection, Sampling and Analysis

Our study drew upon several regional datasets that were used to compile the most exhaustive information as possible on geographical location and total surface area of natural wetlands and farm ponds. Data on geographical location, surface area, density by administrative province, and hydroperiod of 229 natural wetlands were obtained from the Inventory of Wetlands of Andalusia (Inventario de Humedales de Andalucía 2005; Consejería de Medio Ambiente 2005) and additional sources (Morales-Baquero et al. 1999; Casas et al. 2003; Ortega et al. 2003). At present, this is the most comprehensive set of natural wetlands inventoried in Andalusia. All these sources generally adopted a scientific definition of wetlands based on the one given by the National Research Council of USA (National Research Council 1995). According to the Consejería de Medio Ambiente (2005), these wetlands in Andalusia can be classified into four main types (henceforth wetland subtypes) based on criteria of altitudinal location, and geophysical and hydrological characteristics, which roughly summarize their structural and functional diversity: mountain wetlands (MOUNT); wetlands located in the Betic basins and piedmonts (BETBP); wetlands of the Guadalquivir river depression (GUAD); and coastal wetlands (COAST). Many BETBP and GUAD wetlands can be classified as "playa" lakes. Four main hydroperiod length categories were differentiated: permanent; semi-permanent, only drying up during dry years; highly seasonal, always dry during the summer even in rainy years; and ephemeral, flooded only during years with above average precipitation.

Fig. 1 Maps showing the location of the 229 natural wetlands as blue spots (**a**) and 15,578 farm ponds as black dots (**b**) inventoried in Andalusia (southern Spain). Each pixel in (**b**) represents an individual farm pond; larger areas represent multiple individual water bodies, rather than a large, single farm pond. Also is shown (**a**) the location of the eight administrative provinces and the main mountains and Guadalquivir river valley

Table 1 Total surface area and density of waterbody types, natural wetlands (n=229) and farm ponds (n=15,578), by administrative province of Andalusia. Also shown the ratio of total surface area of natural wetlands: farm ponds (AW:AP) and the average annual precipitation in each province

Province	Total surface area of wetlands (AW)(ha)	Total surface area of ponds (AP) (ha)	AW:AP ratio	Density of wetlands (No. 100 km^{-2})	Density of ponds (No. 100 km^{-2})	Average annual precipitation (mm)
Almería	1441	617	2.3	0.20	99.49	193
Cádiz	11851	398	29.8	0.37	7.89	573
Córdoba	217	805	0.3	0.10	7.80	544
Granada	437	254	1.7	0.46	7.42	357
Huelva	49698	1054	47.1	0.37	19.23	500
Jaén	372	783	0.5	0.43	9.31	539
Málaga	1600	201	8.0	0.23	8.76	528
Sevilla	42553	1632	26.1	0.11	9.91	533
Total	108169	5744	18.8	0.28	21.23	471

The inventory of artificial ponds of Andalusia (Casas et al. 2011b) was used to obtain data on farm ponds. A total of 16,544 artificial waterbodies, excluding large reservoirs (>50 ha), was inventoried, of which 15,578 were assigned to farm use as deduced from the predominance of farming activities in the pond environs (Casas et al. 2011b). Data on geographical location, surface area, density by administrative province, potential type of farming activity, and potential construction type were obtained from this inventory. Farming activity was established according to the dominant use in the area, and two main potential construction types were determined in this inventory: natural-looking if pond appeared to have an irregular shape and was connected to the surrounding drainage network, and artificial if the shape was regular (Casas et al. 2011b).

Samples from the totality of waterbodies defined above were used for a more detailed GIS and/or in situ characterization. A sample of 192 natural wetlands was characterized; other wetlands were not taken into account due to different impediments within the available aerial imagery (e.g., cloudiness, snow cover, desiccation). We selected 140 farm ponds after field trips and inspection of pond appearance and accessibility during spring 2007, to represent the variability range of geographical, agronomical, and construction parameters. This inspection revealed four main construction types (henceforth farm pond subtypes): natural-looking ponds were embankments in small streams (EMB) or excavated in natural depressions (EXC), and artificial ones were lined with polyethylene (PET) or concrete (CON). Altitude, surface area, perimeter, % perimeter and average belt-width covered with hydrophytic marginal vegetation, and % land-uses detected in a concentric area within a 2 km distance from the edge of the waterbody (open water+belt of hydrophytic vegetation), were determined by analyzing aerial images (2005) using the geographical information system SIGPAC (2004). Surface area and perimeter of each waterbody were used to estimate shoreline complexity (SC) and a circularity index (CI). SC was calculated as the ratio between shore length (perimeter) and the circumference of a circle with the same area as the wetland. CI was calculated as in Lutton et al. (2010), using the formula $CI=4\pi A/P^2$, where A is the surface area and P is the perimeter of the waterbody. As CI approaches 1 the waterbody surface will be more circular in shape, and as CI approaches 0 the waterbody surface will become more linear in shape.

Chemical characteristics of the 140 farm ponds mentioned above and 95 natural wetlands—a sample representing the four wetland subtypes and their geographical distributions—were determined using water samples taken during May–June 2007. We determined the following variables: pH, electric conductivity (EC_{25}), alkalinity, Cl^-, SO_4^{2-}, Ca^{2+}, Mg^{2+}, Na^+, K^+, dissolved inorganic nitrogen (DIN), total nitrogen (TN), soluble reactive phosphorus (SRP), total phosphorus (TP), chlorophyll *a* of phytoplankton (Chl *a*), and suspended solids (SS). A description of the analytical methods is given in Casas et al. (2011b). Chemical data for natural wetlands were provided by the Consejería de Medio Ambiente, Junta de Andalucía, through the Regional Program of Wetlands Monitoring and Evaluation. Field sampling procedures and water analysis methods were the same for both waterbody types, natural wetlands and farm ponds, and were carried out in the same laboratory (Laboratory for Pollution Surveillance and Control, Junta de Andalucía). During field sampling, the dominant source of water feeding farm ponds was identified by careful inspection of the surrounding area and/or interviewing pond owners or managers of the irrigation systems.

For comparisons of sediment composition, the concentrations of five heavy metals—Cu^{2+}, Fe^{2+}, Mn^{2+}, Pb^{2+} and Zn^{2+}—were available for 71 natural wetlands (data provided by the Consejería de Medio Ambiente, Junta de Andalucía), a sample that represented the four wetland subtypes and their geographical ranges of distribution. We collected sediment samples from 66 farm ponds constituting a subsample of the set of ponds used for water chemical characterization, in which all subtypes and farming systems were represented. All sediment samples were collected in spring 2008 using the same field procedures and analyzed by the same methods in the same laboratory as for water samples. A description of methods for sediment sampling and analysis is given in the website of the Consejería de Medio Ambiente, Junta de Andalucía (http://www.juntadeandalucia.es/medioambiente).

Statistical Analyses

We compared morphological traits and altitude between waterbody types, natural wetlands and farm ponds, using frequency (%) histograms. Surface area was available for 229 natural wetlands and the 15,578 farm ponds inventoried, whereas for other morphological variables the number was reduced to subsets of 192 and 140 waterbodies, respectively. Typological comparisons were carried out on these subsets between waterbody types, natural wetlands vs. farm ponds, and among the eight waterbody subtypes differentiated, for altitude, morphological traits, marginal vegetation traits, and % surrounding land-uses. Comparisons were performed using two-way nested ANOVAs, with the factor subtype nested within the waterbody type factor. Variables were ln (x+1) transformed, except pH, and arcsin $\sqrt{x}$ for percentages, to make the variances homoscedastic (Zar 2010). When significant differences were detected for the factor subtype, Tukey HSD tests for unequal sample size (Zar 2010) were used for post-hoc comparisons between subtype pairs.

Standardized Principal Component Analyses (PCAs) (Legendre and Legendre 1998) on chemical variables of water and sediment were carried out separately to explore multivariate relationships, and summarize and compare variation patterns between natural wetlands and farm ponds. For water variables, average site scores on each of the three main PCs extracted by PCAs and average value of variables with the highest load (>0.75) on each PC for waterbody types and subtypes were compared using two-way nested ANOVAs as above. The same test was used to compare mean concentration of the four heavy metals with high loading (>0.75) on PC1 or PC2 extracted by PCA on sediment data, between waterbody types and among subtypes. All statistical analyses were performed using STATISTICA v7.1 (StatSoft 2005).

Results

On the regional scale total surface area of natural wetlands was almost 19 times higher than that of farm ponds (Table 1). However, the ratio total area of natural wetland to total pond area (AW:AP) varied noticeably. High ratios occur in provinces with high surface area of coastal wetlands (Tables 1 and 2). On the other hand, farm pond area was approximately double that of natural wetlands in the inland provinces (Córdoba and Jaén), and in the most arid eastern provinces (Almería and Granada) pond surface area was around half that of natural wetlands (Table 1, Fig. 1).

As expected from the higher number of ponds inventoried compared to wetlands (Table 2), farm pond density exceeded that of wetlands both at the regional and provincial levels, but it was highly variable (Table 1, Fig. 1). The highest densities were registered in Huelva and, particularly, in Almería where about half the total number of ponds in the region was inventoried. These were mostly small sized, artificial-substrate ponds associated with greenhouse irrigation (Fig. 1, Table 2). Other abundant types were artificial-substrate ponds in woody-herbaceous crops, and natural-looking ponds in the traditional agro-silvo-pastoral “dehesa” system. The latter were particularly abundant in the Sierra Morena range, north of Huelva, Córdoba, and Sevilla, where natural wetlands are scarce (Table 2, Fig. 1).

Generally, natural wetlands showed higher area and perimeter than farm ponds (Fig. 2, Table 3). COAST wetlands had the greatest individual average area, perimeter, and shoreline complexity, and the most elongated shape. GUAD and BETBP wetland types followed in area and perimeter, but with relatively low shoreline complexity, as for artificial-substrate ponds (Table 3). MOUNT wetlands were close in area and perimeter to naturalized-looking ponds, but were more circular in shape and with lower shoreline complexity than EMB ponds. This last was in fact the most elongated waterbody type and with the highest shore complexity after COAST wetlands (Table 3). The two artificial-substrate pond subtypes were the smallest in size and perimeter, particularly CON ponds, and shoreline complexity and circularity index were similar to MOUNT, BETBP, and GUAD wetlands.

Natural wetlands showed on average higher development of marginal vegetation than farm ponds (Table 3). COAST wetlands had the highest development of marginal vegetation (both % perimeter and belt width), followed by BETBP and GUAD types (Table 3). EMB and EXC ponds showed a % perimeter that was fairly vegetated, but much narrower than most natural wetlands. There was scant marginal vegetation development in the two artificial pond subtypes (in most cases it was non-existent), as with most MOUNT wetlands (Table 3).

Farm ponds showed significantly higher % of agricultural land-uses in their environs compared to natural wetlands (Table 3), although EMB ponds showed relatively high % natural vegetation, mainly “dehesa” systems considered as a “semi-natural” vegetation type. However, lowland wetland

Table 2 Total surface area (ha), and number of waterbodies in parentheses, of natural wetlands and farm ponds, by administrative province in Andalusia. The subtypes were differentiated according to geography (natural wetlands), or morphological and land-use criteria using aerial imagery (farm ponds). Subtypes of natural wetlands: MOUNT Mountain, BETBP Betic Basins and Piedmont, GUAD Guadalquivir Depression, COAST Coastal. Subtypes of farm ponds: Naturalized-looking if they appear irregular in shape and connected to the surrounding hydrological network; Artificial if they appear regular in shape and without hydrological connection to their surroundings. Subtypes according to land-use criteria: GRH greenhouse horticulture, WHC woody and herbaceous crops, DHC "dehesa" alone or mixed with herbaceous crops, NV natural vegetation. AW:AP is the ratio; total area of natural wetlands to total pond area

Province	AW:AP	Natural wetlands				Farm ponds						
		MOUNT	BETBP	GUAD	COAST	Naturalized-looking			Artificial			
						WHC	DHC	NV	GRH	WHC	DHC	NV
Almería	2.3	10 (4)	0	0	1431 (14)	5 (3)	0	0	290 (7182)	279 (1438)	0	15 (37)
Cádiz	29.8	0.3 (2)	0.5 (3)	386 (16)	11465 (7)	111 (85)	21 (17)	46 (54)	13 (269)	166 (112)	0	3 (1)
Córdoba	0.3	41 (2)	98 (7)	78 (4)	0	199 (97)	365 (695)	15 (43)	0	155 (255)	6 (6)	1 (3)
Granada	1.7	12 (31)	416 (7)	0	9 (2)	21 (17)	0.1 (1)	3 (9)	3 (58)	159 (789)	3 (11)	3 (16)
Huelva	47.1	0	0	3352 (26)	46346 (12)	177 (246)	369 (1034)	70 (207)	7 (22)	185 (356)	3 (5)	4 (11)
Jaén	0.5	35 (9)	0	337 (50)	0	35 (23)	0	0.4 (4)	0.3 (1)	401 (668)	0.1 (1)	1 (6)
Málaga	8.0	0	1533 (16)	0	67 (1)	15 (22)	0	2 (3)	11 (152)	98 (386)	0	21 (25)
Sevilla	26.1	0	52 (1)	142 (13)	42359 (2)	426 (211)	242 (382)	117 (10)	0.2 (1)	678 (591)	7 (5)	4 (8)
Total region	18.8	98 (48)	2099 (34)	4295 (109)	101677 (38)	989 (704)	997 (2129)	253 (330)	324 (7685)	2121 (4595)	19 (28)	52 (107)
Average area per individual water body		2	62	316	2676	1.40	0.47	0.77	0.04	0.46	0.68	0.48

subtypes (COAST, BETBP, and GUAD) were predominantly surrounded by agricultural land, particularly BETBP and GUAD located in valleys dedicated to agriculture. COAST wetlands showed the highest % urban use, probably due to their location in littoral areas where tourism has proliferated over the last few decades.

Principal Component Analysis (Fig. 3) and nested ANOVAs (Table 4) revealed significant chemical differences between types—natural wetlands and farm ponds—and among waterbody subtypes. PCA extracted three main PCs which accounted for 60% of total variance. PC1 (27% expl. var.) showed high positive loading (>0.8) and an important

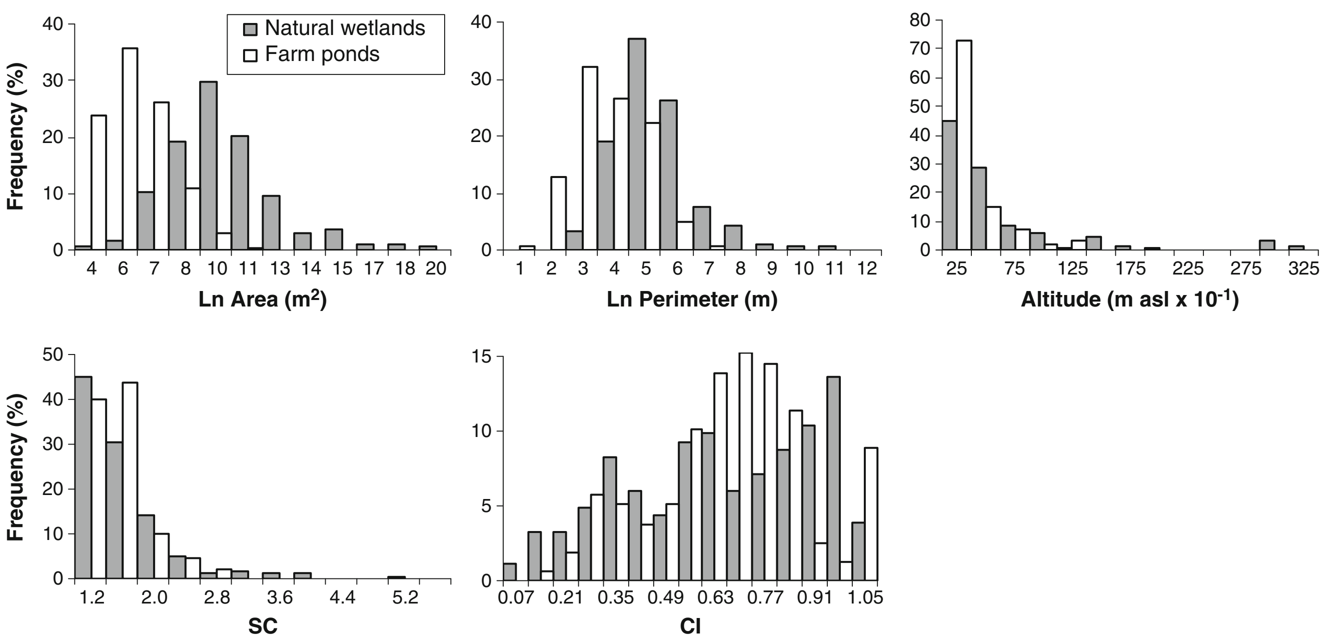

Fig. 2 Frequency (%) of occurrence histograms of waterbodies, natural wetlands and farm ponds, for morphological variables and altitude. Data on surface area correspond to 229 natural wetlands and 15,578 farm ponds. The number of natural wetlands and farm ponds analyzed for other variables are as for Table 3. *SC* shore-length complexity; *CI* circularity index

Table 3 Mean values±1SEM, with range in parentheses, of altitude, morphological characteristics, marginal vegetation (MV) and surrounding land-uses, of the four subtypes of natural wetlands (codes as for Table 2) and the four subtypes of farm ponds identified after field inspections: EMB embankment ponds, EXC excavated ponds, PET polyethylene-lined ponds, CON concrete ponds. The results (F values and level of significance: ns not significant, ***p<0.001) of two-way nested ANOVAs, with waterbody subtype nested within waterbody type, are also shown. Different upper-case letter indicates statistically significant differences (p<0.05) following nested ANOVA and post-hoc Tukey HSD tests, for unequal sample size, comparing subtype pairs

Parameter	Natural wetlands				Farm ponds				ANOVA F values	
	MOUNT	BETBP	GUAD	COAST	Naturalized-looking		Artificial		Factors	
					EMB	EXC	PET	CON	Type	Subtype
N	27	30	99	36	41	19	31	49		
Altitude (m asl)	1924 ± 159^{a} (830–3061)	513 ± 36^{b} (155–830)	281 ± 24^{c} (20–780)	3 ± 0.4^{e} (1–10)	$296\pm30^{b,c}$ (18–735)	$239\pm71^{c,d}$ (19–908)	$310\pm50^{b,c}$ (24–1157)	143 ± 33^{d} (1–1151)	0.6 ns	107.3***
Surface area (ha)	3.4 ± 1.3^{c} (0.04–31.6)	$57.3\pm45.3^{b,c}$ (0.02–1368.2)	123.6 ± 117.3^{b} (0.20–11267.2)	1834.2 ± 1157.1^{a} (0.25–40997.0)	2.6 ± 0.6^{c} (0.2–17.9)	1.6 ± 0.4^{c} (0.06–6.5)	0.5 ± 0.1^{d} (0.02–4.0)	0.04 ± 0.01^{e} (0.005–0.4)	194.1***	33.3***
Perimeter (km)	0.75 ± 0.20^{b} (0.08–4)	1.80 ± 0.63^{b} (0.07–19.3)	1.20 ± 0.12^{b} (0.14–9.12)	7.28 ± 2.25^{a} (0.18–71.78)	0.81 ± 0.11^{b} (0.16–3.24)	$0.51\pm0.25b^{c}$ (0.11–0.99)	0.23 ± 0.03^{c} (0.06–0.80)	0.07 ± 0.006^{d} (0.02–0.29)	169.8***	37.5***
Shoreline complexity	1.24 ± 0.06^{c} (0.87–2.07)	1.22 ± 0.04^{c} (1.00–1.79)	1.30 ± 0.03^{c} (1.00–2.65)	2.27 ± 0.15^{a} (1.13–4.92)	1.59 ± 0.06^{b} (1.00–2.74)	$1.35\pm0.08^{b,c}$ (1.10–2.34)	1.22 ± 0.03^{c} (1.00–1.62)	1.21 ± 0.03^{c} (1.00–2.44)	4.9 ns	33.6***
Circularity index	0.71 ± 0.05^{a} (0.23–1.00)	0.73 ± 0.04^{a} (0.31–1.00)	0.68 ± 0.02^{a} (0.14–1.00)	0.28 ± 0.03^{c} (0.04–0.79)	0.47 ± 0.03^{b} (0.13–0.88)	$0.62\pm0.04^{a,b}$ (0.18–0.82)	0.70 ± 0.03^{a} (0.38–1.00)	0.73 ± 0.03^{a} (0.17–1.00)	3.1 ns	26.2***
% perimeter covered with MV	$12.1\pm6.1^{b,c}$ (0.0–100)	$42.2\pm7.9^{a,b}$ (0–100)	$37.1\pm4.1^{a,b}$ (0–100)	64.2 ± 8.2^{a} (0–100)	28.6 ± 4.3^{b} (0–90)	$54.0\pm9.4^{a,b}$ (0–100)	1.6 ± 1.3^{c} (0–40)	0.6 ± 0.6^{c} (0–30)	17.9***	14.5***
Belt width of MV (m)	$0.4\pm0.2^{c,d}$ (0–6)	15.03 ± 3.11^{b} (0–73)	25.8 ± 5.0^{b} (0–300)	39.4 ± 9.1^{a} (4.3–156)	1.5 ± 0.2^{c} (0–4)	2.6 ± 0.5^{c} (0–8)	$0.2\pm0.1^{c,d}$ (0–2)	0.06 ± 0.05^{d} (0–2)	113.2***	21.5***
% land with natural vegetation	95.4 ± 3.2^{a} (40–100)	$26.6\pm7.8^{b,c}$ (0–100)	$25.7\pm4.1^{b,c}$ (0–100)	$33.1\pm8.3^{b,c}$ (0–100)	42.9 ± 6.9^{b} (0–100)	14.5 ± 7.9^{c} (0–100)	$3.2\pm3.2^{c,d}$ (0–100)	0.3 ± 0.3^{d} (0–15)	50.9***	21.2***
% land with agricultural use	4.6 ± 3.2^{e} (0–60)	$73.4\pm7.8^{a,b,c}$ (0–100)	$71.3\pm4.2^{b,c}$ (0–100)	39.4 ± 8.1^{d} (0–100)	$57.1\pm6.9^{c,d}$ (0–100)	$80.3\pm9.1^{a,b}$ (0–100)	$96.8\pm3.2^{a,b}$ (0–100)	99.5 ± 0.4^{a} (85–100)	68.5***	20.9***
% land with urban use	0 ± 0^{b} (0–0)	0 ± 0^{b} (0–1)	2.9 ± 1.3^{b} (0–80)	25.7 ± 7.1^{a} (0–100)	0 ± 0^{b} (0–0)	5.3 ± 5.3^{b} (0–100)	0 ± 0^{b} (0–0)	0.2 ± 0.2^{b} (0–10)	15.8***	14.0***

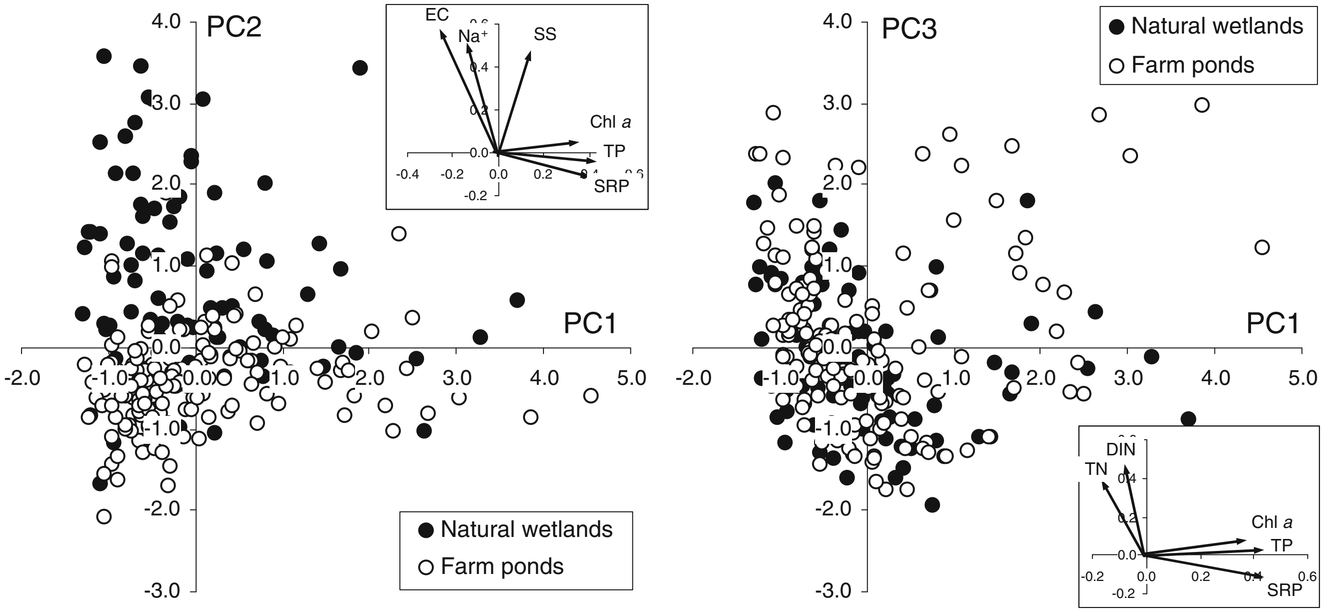

Fig. 3 Results of a standardized Principal Component Analysis on chemical variables of water: biplot PC1 × PC2 on the left, biplot PC1 × PC3 on the right. Symbols in the large plots represent individual waterbodies, differentiating natural wetlands (n=95) and farm ponds (n=140). Small plots on each side represent scores, as vectors, of the environmental variables with loading >0.75 in any PC. PC1—eigenvalue=2.95; PC2—eigenvalue=2.29; PC3—eigenvalue=1.37. *TP* Total phosphorus; *SRP* Soluble reactive phosphorus; Chl *a* Chlorophyll concentration of phytoplankton; *EC* Electric conductivity; *SS* Suspended solids; *DIN* Dissolved inorganic nitrogen; *TN* Total nitrogen

covariation of trophic level variables—TP, Chl *a*, and SRP—but scarcely segregated natural wetlands from farm ponds (Fig. 3). COAST wetlands and PET ponds showed the highest average PC1 score and TP concentration (Table 4). Mineral levels (EC and Na^{+}) together with suspended solids (SS) heavily loaded (>0.75) on positive PC2 (21% expl. var.), where most natural wetlands were clustered, as opposed to most farm ponds arranging on the negative PC2 (Fig. 3). Natural wetlands, with the exception of MOUNT subtype, showed significantly higher scores on PC2 and EC values compared to farm ponds (Table 4). HCO_3^- and Ca^{2+} dominated the ionic composition in all MOUNT wetlands, whereas most COAST were dominated by Cl^- and Na^+ (Table 4). Other wetlands and all pond subtypes showed more diverse ionic composition. Aqueous nitrogen concentration loaded highly (>0.80) on the positive PC3 (12% expl. var.) dimension, where a number of PET and CON ponds were arranged in the most extreme positions (Fig. 3), probably due to their dominant inputs of recycled wastewater and/or groundwater from nitrate-contaminated aquifers (Table 4). Lowland natural wetlands and EXC ponds, surrounded to a large extent by agricultural or livestock activities, showed relatively minor DIN enrichment, which suggests a high magnitude of denitrification processes in their sediments.

Most COAST, and almost half MOUNT, wetlands were permanent waterbodies. Hydroperiod lengths for BETBP and GUAD wetlands were more evenly distributed across the four categories, with seasonal waterbodies being most common (Table 4). All farm ponds can be considered permanent aquatic habitats (J.J. Casas, unpublished data), as they are designed to provide water during periods of scarcity.

The PCA on sediment metals showed a high load of Fe^{2+} (0.90) and Mn^{2+} (0.85) on positive PC1 (44% expl. var.), where most MOUNT and BETBP wetlands, and EMB ponds were arranged (Fig. 4). On average, natural wetlands showed significantly ($F_{(1,129)}$=5.6, p<0.05) higher Fe^{2+} concentration than farm ponds. Cu^{2+} and Zn^{2+} concentrations were highly loaded (0.88) on positive PC2 (31% expl. var.) where most artificial-substrate ponds were arranged. Concentrations of Cu^{2+} and Zn^{2+} were significantly higher in farm ponds compared to natural wetlands ($F_{(1,129)}$=38.9, p<0.001; $F_{(1,129)}$=6.6, p<0.05; respectively). These differences were driven by the PET and CON subtypes (Fig. 4).

Discussion

This study demonstrates that natural wetlands in Andalusia were larger, with higher total area at regional level, but of low density, whilst farm ponds were relatively smaller waterbodies occurring at higher density. The small size of farm ponds suggests their limited value as a habitat for waterbird conservation compared to natural wetlands. Several studies demonstrated for both natural and constructed

Table 4 Mean values±1SEM, with range in parentheses, of site scores on the main PCs extracted by a PCA and the chemical variables of water with the highest load on each PC (see Fig. 3) for the types of natural wetlands and farm ponds. The results (F values and level of significance: ns not significant, $^{**}p<0.01$, $^{***}p<0.001$) of two-way nested ANOVAs, with waterbody subtype nested within waterbody type, are also shown. Different upper-case letter indicates statistically significant differences ($p<0.05$) following post-hoc Tukey HSD tests, for unequal sample size, comparing subtype pairs. The percentage of waterbodies in any group exhibiting certain major anionic and cationic composition of the water is also shown. For each natural wetland subtype the % of waterbodies in four hydroperiod categories is shown: permanent, semi permanent, highly seasonal and ephemeral. Also given for farm ponds is the % of waterbodies in five water-source categories: NDW naturally drained water; CSW canalized surface water; CGW canalized groundwater; RWW, recycled wastewater; RWH rain water harvested

Parameter	Natural wetlands				Farm ponds				ANOVA F values	
					Naturalized-looking		Artificial		Factors	
	MOUNT	BETBP	GUAD	COAST	EMB	EXC	PET	CON	Type	Subtype
N	9	22	43	21	41	19	31	49		
Water chemistry										
PC1 score	1.15±0.16$^{a, b}$ (0.68–2.03)	0.84±0.09$^{a, b}$ (0.09–1.49)	1.33±0.14$^{a, b}$ (0.02–5.00)	1.59±0.32$^{a, b}$ (0.00–4.58)	1.52±0.14^{a} (0.32–3.80)	1.37±0.16$^{a, b}$ (0.39–3.01)	1.54±0.20^{a} (0.12–5.85)	0.85±0.08^{b} (0.03–2.79)	0.4 ns	3.7**
TP (mg L^{-1})	0.15±0.08^{b} (0.00–0.69)	0.12±0.02^{b} (0.02–0.56)	0.32±0.09$^{a, b}$ (0.01–3.80)	0.88±0.25^{a} (0.02–3.23)	0.37±0.10$^{a, b}$ (0.02–2.42)	0.23±0.07$^{a, b}$ (0.01–1.27)	0.66±0.17^{a} (0.01–4.71)	0.13±0.05^{b} (0.01–2.44)	0.0 ns	4.7***
PC2 score	1.80±0.24^{b} (0.92–2.57)	3.15±0.29^{a} (1.23–5.64)	2.61±0.18^{a} (0.41–5.51)	2.95±0.20^{a} (1.05–4.67)	1.75±0.14^{b} (0.38–3.97)	1.93±0.14$^{a, b}$ (0.92–3.19)	1.60±0.09^{b} (0.00–3.46)	1.60±0.05^{b} (0.77–2.57)	54.2***	3.6**
EC (mS cm^{-1})	0.27±0.09^{c} (0.02–0.70)	34.78±13.05^{a} (0.14–230.00)	15.54±4.70^{a} (0.04–146.30)	25.49±6.68^{a} (0.66–120.10)	1.79±0.61^{b} (0.08–24.00)	2.33±0.58^{b} (0.26–9.65)	2.10±0.27^{b} (0.23–9.37)	1.79±0.16^{b} (0.44–4.75)	35.5***	8.7***
PC3 score	1.11±0.24^{c} (0.46–2.64)	1.77±0.17^{b} (0.35–3.95)	1.83±0.13^{b} (0.00–3.75)	2.02±0.20$^{a, b}$ (0.79–3.75)	1.43±0.15$^{b, c}$ (0.19–4.56)	1.67±0.19^{b} (0.20–3.09)	2.46±0.15^{a} (0.61–4.92)	2.31±0.14$^{a, b}$ (0.78–4.82)	4.4 ns	6.7***
DIN (mg L^{-1})	0.31±0.25^{c} (0.01–2.32)	0.55±0.18^{c} (0.01–3.31)	0.71±0.20^{c} (0.01–6.18)	0.46±0.18^{c} (0.00–3.77)	1.27±0.48^{b} (0.03–13.69)	0.42±0.09^{c} (0.02–1.25)	3.98±0.92^{a} (0.04–29.42)	2.46±0.57$^{a, b}$ (0.03–18.39)	15.0***	4.7***
Major ions										
HCO_3^-	100	9	33	14	59	42	33	47		
SO_4^{2-}	0	32	16	10	17	26	27	37		
Cl^-	0	59	51	76	24	32	40	16		
Ca^{2+}	100	18	26	5	37	42	38	37		
Mg^{2+}	0	18	12	5	15	5	13	43		
Na^+	0	64	63	90	49	53	50	20		
§Hydroperiod (natural wetlands) or †water source (farm ponds)										
§Permanent †NDW	45	17	8	80	100	37	0	0		
§Semi-permanent †CSW	15	21	25	9	0	16	29	16		
§Highly seasonal †CGW	33	45	38	9	0	42	45	79		
§Ephemeral †RWW	7	17	29	2	0	5	22	2		
†RWH					0	0	3	2		

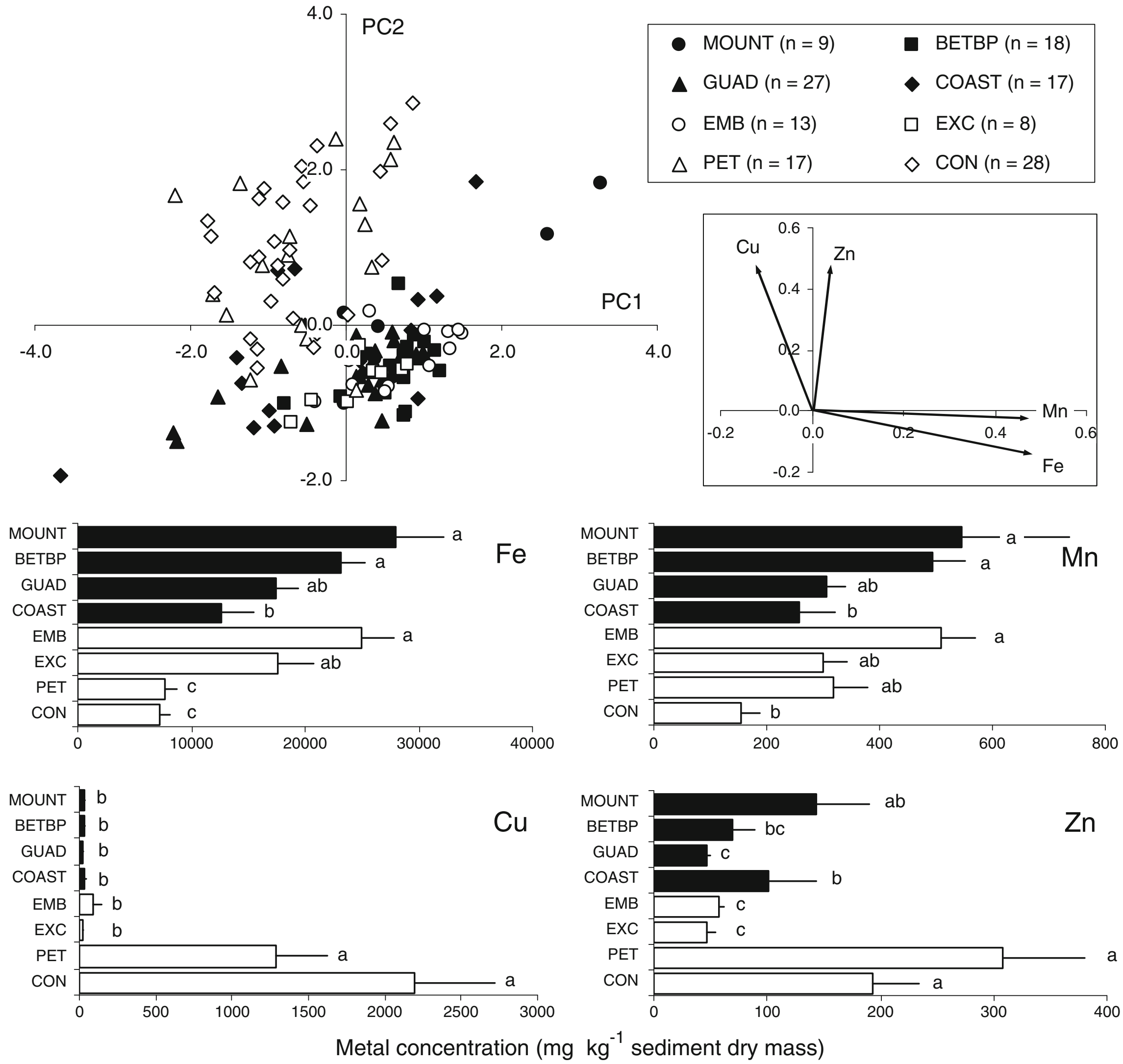

Fig. 4 Biplot PC1 × PC2, upper part, results of a standardized Principal Component Analysis on the concentration of metals in sediment. Symbols represent different waterbody subtypes, solid symbols are natural wetlands (n=71) and open symbols farm pond (n=66). The small plot on the right represent scores, as vectors, of the four metals with loading >0.75 on any PC. The four plots below represent the mean (+1 SEM) concentration in sediment of the four metals with high load on PC1 or PC2 for each waterbody subtype: black columns represent natural wetlands and white columns farm pond. Different letter indicates statistically significant differences (p<0.05) following nested ANOVA and post-hoc Tukey's test, for unequal sample size, comparing subtype pairs

wetlands, that larger waterbodies are of greater conservation value than smaller ones in that they support more diverse waterbird assemblages (Froneman et al. 2001; Paracuellos and Telleria 2004; Sánchez-Zapata et al. 2005). However, Froneman et al. (2001) also suggested that a large number of farm ponds can ensure a high diversity of physical habitat attributes, which is essential in supporting a large diversity of waterbirds at regional scales. Furthermore, small waterbodies can play a major role for certain wetland taxa that do not require large waterbodies to thrive (Gibbs 1993; Oertli et al. 2002; Thiere et al. 2009).

The relative total surface area of natural wetlands to farm ponds in Andalusia can be considered low. The ratio area of natural wetlands to area of farm ponds (AW:AP) measured for this region was less than half, much lower still in the continental and most arid provinces, compared to the global ratio of 55 given by Downing et al. (2006). This can be attributed to relatively low natural wetland abundance due to dominant semiarid climate and to the severe wetland loss over the last decades (Casado and Montes 1995), but also to the recent proliferation of farm ponds in the highly profitable irrigated areas (Casas et al. 2011b).

The relatively high farm-pond density might have important implications for biodiversity conservation in Andalusia. In some provinces farm pond density almost equals the density of natural wetlands reported for some humid temperate regions (see Gibbs 2000). Wetland density is a crucial factor for sustaining many species with short dispersal distances, such as amphibians, that live in multiple, local populations sustained through occasional migration (i.e., metapopulations) (Gibbs 2000). Therefore, in areas of high pond density farm ponds could play a prominent role in the metapopulation dynamics of these species, serving as "stepping stones" for dispersal across the landscape. This seems particularly true in "dehesa" systems where ponds are embedded in a semi-natural landscape matrix. However, in some areas of high pond density, intensive agricultural uses might hinder animal dispersion, such as in south-east and south-west areas dedicated to greenhouse or open-air intensive crops, respectively. In fact, a recent survey carried out on 46 farm ponds in Andalusia reported 12 (5 on the endangered list for Spain) of the 18 amphibian species known in this region, primarily in naturalized ponds linked to "dehesa" systems (Peñalver et al. 2010). Studies in fragmented landscapes indicate that local amphibian occurrence and species richness tend to decline with increasing isolation from other ponds and conversion of land to intensive uses, such as agriculture and roads (Gibbs 1998a,b; Richter-Boix et al. 2007). Wetland density is often assumed to be of less importance for metapopulation dynamics of species with ample, active or passive, dispersal abilities, such as many micro and macroinvertebrate species. Nevertheless, Thiere et al. (2009) demonstrated that density of constructed wetlands in agricultural landscapes can also enhance α and γ diversities of benthic macroinvertebrates.

Littoral zones of waterbodies are widely recognized to support a rich biota, often comprising much of the whole system biodiversity (reviewed by Strayer and Findlay 2010). Shore structural complexity provides refuge against predation or wave action and a diverse array of microhabitats for small organisms (Schmude et al. 1998; Hansson et al. 2005). In our study, COAST wetlands showed the highest shoreline complexity and the lowest circularity index, followed by EMB and EXC ponds. This suggests a high potential for biodiversity conservation of littoral zones in these waterbody subtypes compared to artificial-substrate ponds and even many natural wetlands. The low complexity of perimeter of BETBP and GUAD wetlands might be largely attributed to high agricultural pressure. Many wetlands in these subtypes are devoid of marginal vegetation, and thus have shore habitats that are highly exposed to sedimentation from tillage activities. The two artificial pond subtypes—CON and PET—also showed low shoreline complexity, which, together with the high slope angle of the margins (ranging between 45–90 degrees; Casas et al. 2011b) and artificial substrate, would limit the potential of their littoral zones to support biodiversity. A lack of marginal vegetation can have a negative effect on insect species dependent on shoreline vegetation as adults (Butler and DeMaynadier 2008).

The hydrologic regime strongly influences abundance and composition of littoral-zone biota (reviewed by Strayer and Findlay 2010), and the whole waterbody diversity in shallow systems when drawdown is extreme. Although we lack specific data on the depths of natural wetlands, most were shallow, with maximum depths of 2 m or less and marked water level fluctuations due to the high evaporation and seasonality of rainfall in the region (Consejería de Medio Ambiente 2005). This is particularly true for wetlands of the GUAD and BETBP types, generally classified as "playa" lakes characterized by high seasonal and interannual water level fluctuations (Rodríguez-Rodríguez et al. 2006). Farm ponds are, broadly speaking, permanent waterbodies, due to their agronomical function to supply water during dry periods. Therefore farm ponds have the potential to enlarge the hydroperiod gradient at the regional level, particularly in lowland agricultural inland regions that are dominated by temporary natural wetlands. Several studies have concluded that maintaining wetlands with a high hydroperiod diversity in a landscape benefits diversity of amphibians (Beja and Alcazar 2003; Babbitt 2005; Richter-Boix et al. 2006) or enhance regional invertebrate biodiversity (Serrano and Fahd 2005; Tarr et al. 2005; Waterkeyn et al. 2008). However, the artificial hydrology of farm ponds might hinder native species adapted to the natural hydrologic regimes in the region and favor exotic species. Bunn and Arthington (2002) suggested that an artificially altered hydrological regime might favor non-native species. This is most likely in PET and CON ponds where their small size, disconnection from natural runoff, and primary function for irrigation accentuates artificial water-level fluctuations compared to EXC and EMB ponds. Moreover, the permanent hydrology in farm ponds might have drawbacks for the conservation of native species, such as amphibians. Many amphibians cannot tolerate the presence of predatory fish in their breeding sites (Smith et al. 1999), which is most likely in permanent habitats. For instance, numerous farm ponds, as well as permanent natural wetlands, in Andalusia support exotic mosquitofish (*Gambusia* spp.) and crayfish (Peñalver et al. 2010), two taxa with potentially negative effects on amphibians (Kats and Ferrer 2003).

Variation in regional salinity values can also be a factor to consider for the enhancement of biodiversity at the regional level, due to the strong selective effect of salinity on wetland community composition (Boix et al. 2007; Waterkeyn et al. 2008). Given that farm pond salinity is generally much lower than natural wetlands in lowland areas, together with a diverse ionic composition, our findings have implications for conservation. A recent study on microinvertebrates—rotifers and crustaceans—demonstrated that farm ponds in Andalusia harbor rich assemblages of freshwater taxa not recorded in natural wetlands, which in turn harbor halotolerant or halophilous taxa not found in farm ponds (León et al. 2010). High salinity in wetlands is primarily determined by natural factors: evaporative concentration, endorheism, or basin lithology in BETBP and GUAD wetlands (Rodríguez-Rodríguez et al. 2006), or intrusion of sea-water in COAST wetlands. Additionally, as the hydrology of most wetlands in Andalusia is highly dependent on groundwater inputs (Reques 2006), it is to be expected that aquifer overexploitation might lead, apart from hydroperiod shortening, to secondary salinization by modification of water balance in inland wetlands (Rodríguez-Rodríguez et al. 2006), or by enhancing sea-water intrusion in coastal wetlands. Furthermore, we must consider that many wetlands lost in Andalusia during the past were oligohaline, due to agricultural reclamation of fertile arable land (Reques 2006). As concluded by Waterkeyn et al. (2008), it is to be expected that the combined and interacting influences of salinization and hydrological impairment of Mediterranean wetlands will potentially induce a considerable decline in regional diversity. This underpins the value for biodiversity conservation of the permanent-oligohaline habitats provided by farm ponds.

Eutrophication leads to homogenization of biotic communities and jeopardizes the conservation of many intolerant, often endangered, species (Smith 2003). For instance, Boix et al. (2007) reported that eutrophic-hypertrophic waterbodies were among the wetlands types with the lowest species richness in a Mediterranean region. Natural wetlands and farm ponds in Andalusia were not fundamentally different in their trophic status, but important variation was detected within and between subtypes. Coastal wetlands and PET ponds were, on average, the most eutrophic waterbodies, likely due to intensive agricultural and urban land uses in the surrounding areas and the influx of recycled wastewaters, respectively. In contrast, many BETBP wetlands and CON ponds were oligotrophic, possibly due to high salinity that might buffer eutrophication and above-ground positions that prevented the reception of agricultural drainages, respectively. One possible implication of these results is that in areas where natural wetlands are generally eutrophic, non-eutrophic farm ponds might play an important role for aquatic biodiversity conservation. For example, the bulk of the population of the endangered Iberian toothcarp (*Aphanius iberus*) in eastern Andalusia was concentrated in irrigation CON ponds with submerged aquatic vegetation, whereas natural coastal wetlands with chronic eutrophication supported impoverished populations (Casas et al. 2011a).

Extremely high copper concentration in sediments was common in artificial-substrate ponds in intensive production systems. This can be attributed to the frequent usage of copper sulphate as a biocide to control algae and submerged aquatic vegetation (Casas et al. 2011a, b). The copper in these ponds may limit resident biodiversity.

Studies in a diversity of climatic regions (Froneman et al. 2001; Abellán et al. 2006; Markwell and Fellows 2008; Lutton et al. 2010) generally conclude that the primary function of farm ponds for water supply and their management and construction characteristics make them poorly suited to replace natural wetlands. Our findings are in general agreement with this conclusion because natural wetlands in Andalusia are fundamentally different from farm ponds. However, farm ponds may play a role in region-wide habitat complementarity to natural waterbodies. Farm ponds provide a high density of small oligohaline permanent waterbodies not provided by natural wetlands, although some construction and management features of artificial-substrate ponds in intensive farming systems probably hinder development of their ecological potential. In Andalusia, no program of wetland creation for nutrient retention and biodiversity conservation in agricultural landscapes currently exists. Improving farm pond construction and management to better match agricultural function with biodiversity conservation might to be a low-cost option, although it would require collaboration from growers within a technical and policy framework which, at present, is still lacking.

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