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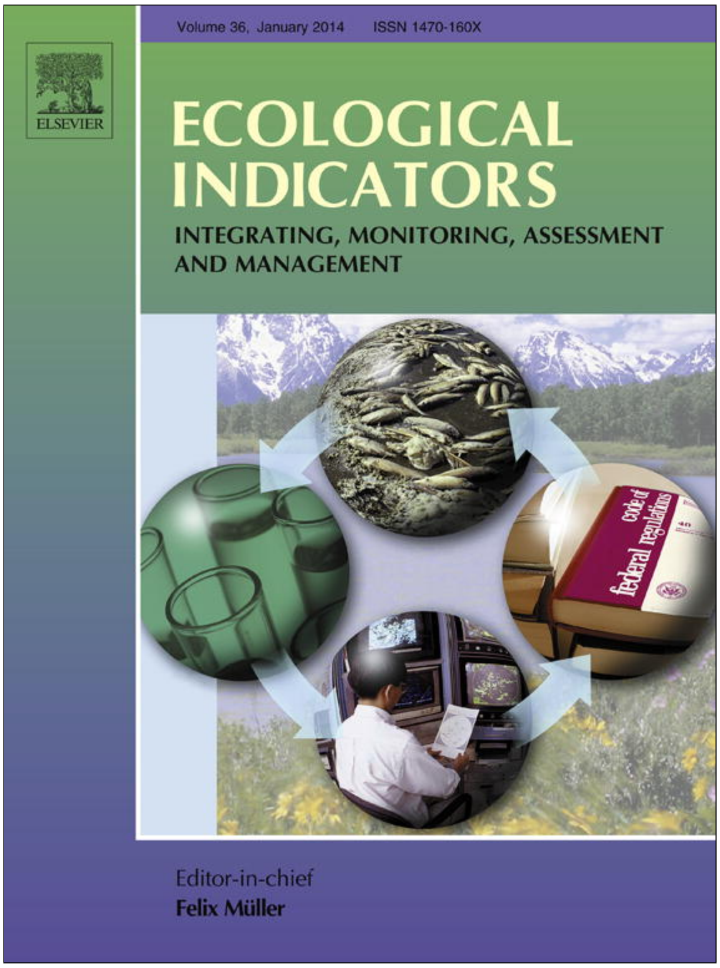

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Ecological Indicators 36 (2014) 400–408

Contents lists available at ScienceDirect

Ecological Indicators

journal homepage: www.elsevier.com/locate/ecolind

Original articles

Wetland and landscape indices for assessing the condition of semiarid Mediterranean saline wetlands under agricultural hydrological pressures

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A R T I C L E I N F O

Article history:
Received 3 April 2013
Received in revised form 19 July 2013
Accepted 9 August 2013

Keywords:
Semiarid wetlands
Remote sensing
Irrigated agricultural areas
Plant taxa
Watershed hydrological condition
Wetland condition

A B S T R A C T

During last decades semiarid Mediterranean saline wetlands have undergone several hydrological and biological changes as a consequence of increased water inputs from agricultural areas. Specific indices are needed in order to assess the condition of these unique ecosystems in relation to major hydrological disturbances at watershed level. Through the long-term study of selected plant taxa in a set of representative wetlands in Murcia province (SE Spain), together with the characterization of their watershed agricultural land uses, plant indicators of wetland condition were sought and then combined into a wetland condition index. Percentages of land cover classes of interest were weighted taking into account land cover arrangement and receiving wetland size. Characteristic perennial plant taxa were sampled in 1989 and 2008 and significant taxa frequency changes at each wetland site were determined. Regression analysis was used to relate wetland plant taxa frequency and watershed condition during the study period. *Limonium* spp., *Arthrocnemum glaucum*, *Phragmites australis*, *Tamarix canariensis* and *Atriplex halimus* showed significant relationships with watershed condition. Indicator taxa were thus identified and their frequencies were combined into an integrated index of wetland condition, which showed a robust relationship with watershed hydrological condition.

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1. Introduction

Wetlands naturally act as sinks of surface and subsurface drainage effluents due to their relative low position in the landscape, thus reflecting upland occurring processes. In last decades, specially in coastal plain areas, there has been a flourishing growth of agricultural irrigated land areas in many semiarid Mediterranean regions (Herrero and Snyder, 1997; Levin et al., 2009; Martínez-Fernández et al., 2005). More specifically, in Murcia province the opening of the Tagus-Segura river water transfer in 1979 promoted the conversion of most dry farmed lands into irrigated land areas. The current expansion of agricultural irrigated lands at watershed scale has led to significant hydrological imbalances that alter wetland ecosystems, thus threatening its conservation (Casta neda and Herrero, 2008a; Esteve et al., 2008; Ortega et al., 2004).

Monitoring actions applied to management and conservation of wetlands require precise indicators in order to obtain accurate information about ecosystem state and functioning and provide an effective early warning system (Fancy et al., 2009). Although much effort has been applied towards protection of wetlands, the preservation of surrounding areas in which they are embedded has been ignored (Houlahan and Findlay, 2004). Assessment of watershed hydrological condition plays therefore a vital role in wetland management (Mack, 2006; Turner et al., 2003; Wigand et al., 1999).

The European Water Framework Directive (European Commission, 2000) seeks the development of indicators of ecological status for freshwater ecosystems, specifically including wetlands (European Commission, 2003). However, most indicators established for aquatic ecosystems are not suitable for semiarid Mediterranean saline wetlands, also called crypto-wetlands (Carre no et al., 2008; Innis et al., 2000; Williams, 1999). These are semi-terrestrial environments between steppes and standing water ecosystems (Vidal-Abarca et al., 2003; Casta neda and Herrero, 2008b).

The development and selection of ecological indicators is a complex process for which different approaches can be used (Carignan and Villard, 2002; Niemeijer and Groot, 2008). Physico-chemical indicators of wetland habitat condition can be very labor intensive and may not necessarily be biologically relevant (Gergel et al., 2002). However, biotic indicators, and specifically plants, may integrate changes in wetland condition over time, may be easy to identify and may provide information on the type of pressures if

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http://dx.doi.org/10.1016/j.ecolind.2013.08.007

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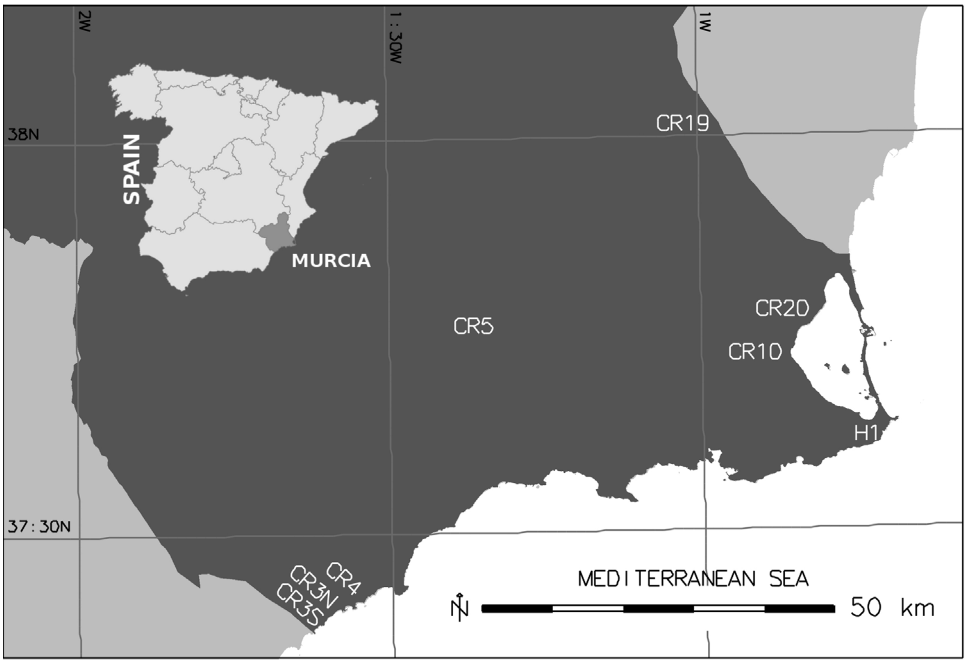

Fig. 1. Location map of the study wetlands in Murcia province. Wetland keys: Cañada Brusca North (CR3N) and South (CR3S), Alcanara (CR5), Marina del Carmolí (CR10), Playa de la Hita (CR20), Matalentisco (CR4), Boquera de Tabala (CR19) and Salinas del Rasall (H1).

their ecological tolerances are known (Cronk and Fennessy, 2001; Mitsch and Gosselink, 2007).

Plant species diversity of semiarid saline wetlands is relatively low and it is differentiated according to the plants' individual tolerance of salinity, fluctuating water table levels and anoxic substrate. The establishment of plants as ecological indicators comprises the study of species responses to a known range of a given environmental stressor (Allan, 2004; Niemi and McDonald, 2004). In this regard, results arising from the study of wetland plant species and surrounding land uses at single dates may not be representative of the whole range of species responses and environmental gradients. It has been suggested, for example, that there is a delayed response of the biota to certain landscape environmental variables (Carre no et al., 2008; Harding et al., 1998). Besides, species might still occur in wetlands long after the conditions that promoted their presence have vanished. Therefore, interpretation of site and date specific data needs to be conducted within the larger spatial and temporal perspective (Álvarez-Rogel et al., 2006; Dale and Beyeler, 2001).

Although several indices based on plant species and communities have previously been used as a tool for wetland condition assessment in the USA (Johnston et al., 2009; López and Fennessy, 2002; Miller et al., 2006), such indices are lacking for semiarid Mediterranean saline wetlands and are needed in order to fulfill the European Water Framework. In the context of a proposal for monitoring semiarid Mediterranean saline wetlands, the main aim of this study was to investigate plant taxa as an ecosystem attribute that reflects long-term changes in wetland hydrological conditions.

More specifically, the objectives were (1) to assess major changes in wetland plant taxa composition and in their associated watershed land cover classes in a set of representative semiarid saline wetlands over a 20 years period, (2) to characterize watershed hydrological condition for each wetland in relation to agricultural hydrological pressures, (3) to establish relationships between wetland plant taxa and watershed hydrological condition, and (4) to develop and validate a wetland condition index based on the resulting indicator plant taxa. To accomplish this, historical fieldwork sampling, remote sensing and hydrological modeling techniques were combined.

2. Methods

2.1. Study wetlands

Murcia province (SE Spain: 37° N, 1° W) has a semiarid Mediterranean climate with a mean annual temperature of 16 °C and a mean annual precipitation of 339 mm (Esteve et al., 2006). Eight representative wetlands from which we had plant records in 1989 and 2008 were selected, *i.e.* 6 coastal and 2 inland wetlands (Fig. 1 and Table 1). Selected sites are included in the regional inventory of wetlands (Vidal-Abarca et al., 2003) and their protection status ranges from regional, national to international level due to their high ecological values (Ramsar Site, Special Protection Area for Birds, Site of Community Importance and Special Protection Area for the Mediterranean), except for Matalentisco and Boquera de Tabala wetlands, which do not benefit from any protection status. Marina del Carmolí and Playa de la Hita wetlands are in a lowland coastal plain, called Campo de Cartagena, associated with the internal shore of the Mar Menor coastal lagoon, which comprises 12,700 ha (Conesa, 1990; Conesa and Jiménez-Cárceles, 2007). The lagoon and its associated wetlands are all RAMSAR sites, containing

Table 1
Characterization of wetlands and their associated watersheds.

Name	Wetland size (ha)	Watershed size (ha)	Location	Reference site
Salinas del Rasall	26.3	236	Coastal	Yes
Cañada Brusca South	3.8	69.5	Coastal	Yes
Cañada Brusca North	17.4	360	coastal	No
Marina del Carmolí	314	16,923	Coastal	No
Playa de la Hita	34.4	2052.8	Coastal	No
Matalentisco	10.4	907.6	Coastal	No
Boquera de Tabala	36.9	5819.2	Inland	No
Alcanara	199	6508	Inland	No

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eighteen Habitats of Community Interest according to the European Habitat Directive (Council of Europe, 1992). Salinas del Rasall is a coastal wetland associated with a salt extraction pond embedded in the Calblanque Natural Park, protected in 1987 by the regional authorities and also included in the Mar Menor RAMSAR protected area. Matalentisco and Cañada Brusca North and South are coastal wetlands located in the southern part of the region on the Mediterranean Sea. Boquera de Tabala and Alcanara are inland wetlands associated with an ephemeral river and with a saline alluvial plain, respectively. Salinas del Rasall and Cañada Brusca South wetlands were selected as reference sites, as their watersheds did not present irrigated agricultural land areas during the study period.

Characteristic plant communities of these wetlands are salt steppes and salt marshes, which occupy areas with high salinity conditions and low and high water availability, respectively. The salt steppe units are composed of the priority habitat 1510 "Mediterranean salt steppes, Limonietalia" and habitat 1430 (Halo-nitrophilous scrubs Pegano-Salsoletea) of the European Habitat Directive (European Commission - Directorate General for Environment, 2007). Main species in salt steppe are *Lygeum spartum, Suaeda vera, Frankenia corymbosa* and *Limonium* spp. The salt marsh units are dominated by habitat 1420 (Mediterranean and thermo-Atlantic halophilous scrubs, Sarcocornetea fruticosi) and habitat 1410 (Mediterranean salt meadows). Main species in saltmarsh are *Sarcocornia fruticosa, Arthrocnemum glaucum, Halimione portulacoides and Halocnemum strobilaceum*. Habitat 92D0 (Southern riparian galleries and thickets) is also represented in sandy areas, composed of *Tamarix canariensis* and *Tamarix boveana*. Finally, the reed beds unit, when present, is dominated by *Phragmites australis* and occupies areas with regular water flows and lower salinity (Álvarez-Rogel et al., 2006, 2007; MARM, 2009). All habitats, excluding reed beds, are recognized as being of Community Interest by the Habitat Directive and habitat 1510 (Mediterranean salt steppes) is designated of Priority Interest.

2.2. Watershed delineation

Specific watershed boundaries for each wetland were delineated from a 10 m raster digital elevation model (DEM) using single flow direction method (D8 algorithm; O'Callaghan and Mark, 1984). Prior to watershed delineation, the DEM was modified in the Campo de Cartagena coastal plain area by lowering the elevation values coinciding with existent stream network to force flow-direction models to match existing stream lines (King et al., 2005; Strayer et al., 2003). Since watersheds are usually delineated by the area upstream from a given outlet point, DEM elevation values within larger wetlands in the Campo de Cartagena area were modified by creating an artificial sink in order to force all flow-accumulation cells draining into the wetland to converge into a single cell. All GIS analyses were performed with GRASS GIS 6.4 (GRASS Development Team, 2008). For more details, see Martínez-López et al. (2012). Wetland watersheds ranged from 70 to 17,000 ha (Table 1).

2.3. Land use/land cover mapping

Historical and medium spatial resolution land cover data were needed to assess specific pressures on the study wetlands over time. For each wetland watershed, land cover maps in years 1987 and 2008 were obtained by means of remote sensing techniques. Supervised classification of eleven land cover classes was performed using the widely used maximum likelihood algorithm (Michelson et al., 2000; Richards and Jia, 2006). Landsat images sensors TM and ETM+were used for years 1987 and 2008, respectively, and image pixel size was set to 25 m. Each classification was carried out with two images (summer and winter), including single bands and diverse spectral and patch shape indices as ancillary layers: Normalized Difference Vegetation Index (NDVI; Bannari et al., 1995), Modified Normalized Water Index (MNDWI; Hui et al., 2008), Normalized Difference Built-up Index (NDBI; Zha et al., 2003) and Normalized Difference Bareness Index (NDBaI; Chen et al., 2006), as well as patch fractal dimension (FD) and patch shape indices (SI) (Chust and Ducrot, 2004). Land use patches were previously extracted by means of automatic image segmentation using the region growing algorithm with SPRING software (Cãmara et al., 1996). Training sites were obtained using aerial photos from 1987 and 2008 of 1 m and 0.45 m resolution, respectively. The classification method was enhanced by an iterative training sites random selection method (Carre no et al., 2011; González, 2011). The resulting maps were verified by visual validation on aerial photos using a stratified random sampling. For more details, see Martínez-López et al. (2012).

Seven land cover classes were studied: dense and open natural woodlands, dense and open natural shrublands, rainfed tree and herbaceous croplands, irrigated tree and herbaceous croplands, greenhouses, urban areas and water bodies. Urban areas and water bodies were not further considered in subsequent analyses, since they represented low percentages in the studied watersheds. Rainfed cropland areas were also disregarded since they were historically present and their water outputs are negligible in comparison to those from irrigated cropland areas (Conesa, 1990; Velasco et al., 2006). The rest of land cover classes were pooled into two categories: natural areas (NAT), including dense/open natural woodlands and dense/open natural shrublands, and irrigated agricultural land areas (ILA), including irrigated tree and herbaceous croplands and greenhouses. Land cover maps showed an overall accuracy percentage of 73% and 83% for 1987 and 2008 classification, respectively. There was a significant and a marginally significant inverse Pearson correlation between natural and irrigated land areas in wetland watersheds, being higher in 2008 ($r = -0.83$; $P = 0.01$) than in 1989 ($r = -0.69$; $P = 0.06$).

2.4. Watershed hydrological condition index

In order to assess and compare the hydrological pressures that irrigation at watershed scale exerts on different wetlands, raw percentages of irrigated and natural land cover classes in the watersheds were weighted by landscape factors. First, since the irrigation flows from near irrigated areas were supposed to exert more influence on wetland hydrology (Casta neda and Herrero, 2008b), we used inverse-distance weights (IDW) to characterize land-cover arrangement within watersheds (King et al., 2005; Van Sickle and Johnson, 2008). Secondly, since wetland sizes differed over two orders of magnitude, ranging from 3.8 to 314 ha, we considered that the effect of drainage inputs on larger wetlands should be considerably lower than in smaller ones. Thus, IDW percentages of natural and irrigated land areas in the watershed were divided by the receiving wetland area and then square rooted, according to a wetland area relative percentage index (WARP; Martínez-López et al., 2014; Eq. (1)).

$$\mathrm{WARP}_{LC} = \sqrt{\frac{\mathrm{LC\ (IDW)\ in\ watershed\ (\%)}}{\mathrm{Wetland\ area\ (ha)}}} \qquad (1)$$

where LC refers to a specific land cover of interest (natural and irrigated land areas), which were first inverse-distance weighted (IDW).

2.5. Vegetation sampling

To characterize plant taxa composition at each wetland site, frequency of nine dominant characteristic perennial plant taxa was recorded in years 1989 and 2008 by means of field sampling.

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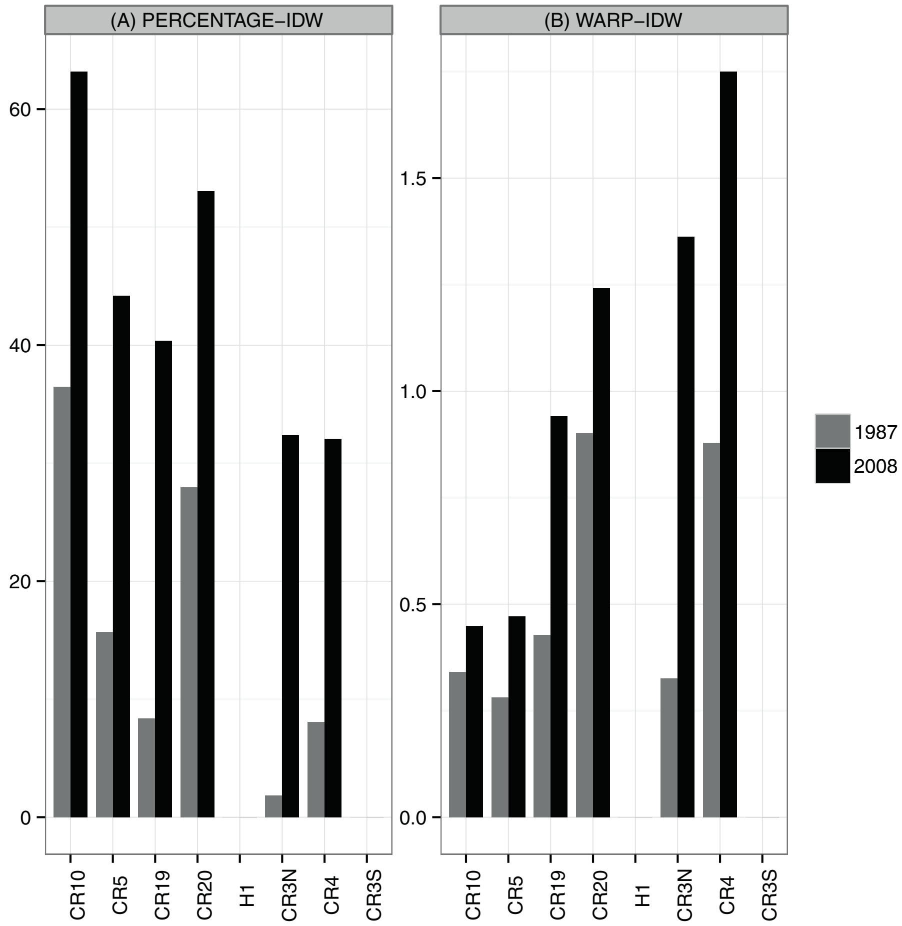

Fig. 2. Irrigated agricultural land areas (ILA) in wetland watersheds in 1987 and 2008. Panels A and B show the percentage of irrigated land areas (inverse distance weighted: IDW) and the wetland area relative percentages (WARP-IDW), respectively. Wetland keys: Cañada Brusca North (CR3N) and South (CR3S), Alcanara (CR5), Marina del Carmolí (CR10), Playa de la Hita (CR20), Matalentisco (CR4), Boquera de Tabala (CR19) and Salinas del Rasall (H1). Wetlands are arranged from larger to smaller sizes from left to right.

Sampled taxa were: *A. glaucum*, *Atriplex glauca*, *Atriplex halimus*, *Limonium* spp. (including *Limonium cossonianum*, *Limonium caesium* and *Limonium insigne*), *T. canariensis*, *P. australis*, *Suaeda vera*, *Sarcocornia fruticosa* and *Halimione portulacoides*. The number of sampling units in each wetland ranged from 12 to 116, depending on wetland size and heterogeneity of plant communities. Sampling units consisted of 25 m^2 circular areas regularly distributed. Presence/absence of selected taxa was recorded at each sampling unit and overall taxa frequencies, ranging from zero to one, were then calculated for each wetland. Wetland areas remained constant throughout the study period except for Boquera de Tabala wetland, which was slightly reduced in size during the study period due to road works.

2.6. Selection of indicator plant taxa

Taxa showing significant frequency changes at each wetland site over the study period were first identified using a pairwise comparison binomial test (Crawley, 2007) in order to select potential indicator taxa, whose frequencies might be linked to watershed hydrological condition. Subsequently, final indicator taxa were selected by means of linear regression analyses between their frequency and watershed hydrological condition in years 1989, 2008, and the observed changes during the study period. Taxa frequencies were arcsine square root transformed to accomplish normality prior to regression analysis (Logan, 2010). Careful inspection of residual plots was performed in order to check for normality and independence of residuals. Data were analysed with the statistical package R (R Core Team, 2012).

2.7. Wetland condition index

Frequencies of resulting indicator taxa were combined as an index to assess wetland condition (WCI) in relation to watershed hydrological pressures. For this purpose, the sum of taxa frequencies that were positively related to hydrological alterations (considered as negative indicator taxa) was substracted to the sum of taxa frequencies that were positively related to naturalness (considered as positive indicator taxa). Total frequency of positive indicator taxa was square rooted in order to increase their weight at low occurring frequencies, while diminishing frequency values higher than one. The fact that both positive indicator taxa were occurring at high frequencies, could probably be more related to wetland natural heterogeneity than to its condition status. On the contrary, the sum of negative indicator taxa was squared in order to underpin the presence of more than one negative indicator taxon (sum of values higher than one), while minimizing the effect of total frequencies below one (Eq. (2)).

$$WCI = \sqrt{\text{Total freq. positive indicator taxa}} - (\text{Total freq. negative indicator taxa})^2 \quad (2)$$

In order to test the applicability of the proposed index in a wider set of wetlands, four additional wetlands (Sombrerico, Lo Poyo, Ajauque and Derramadores) were selected in Murcia province, for

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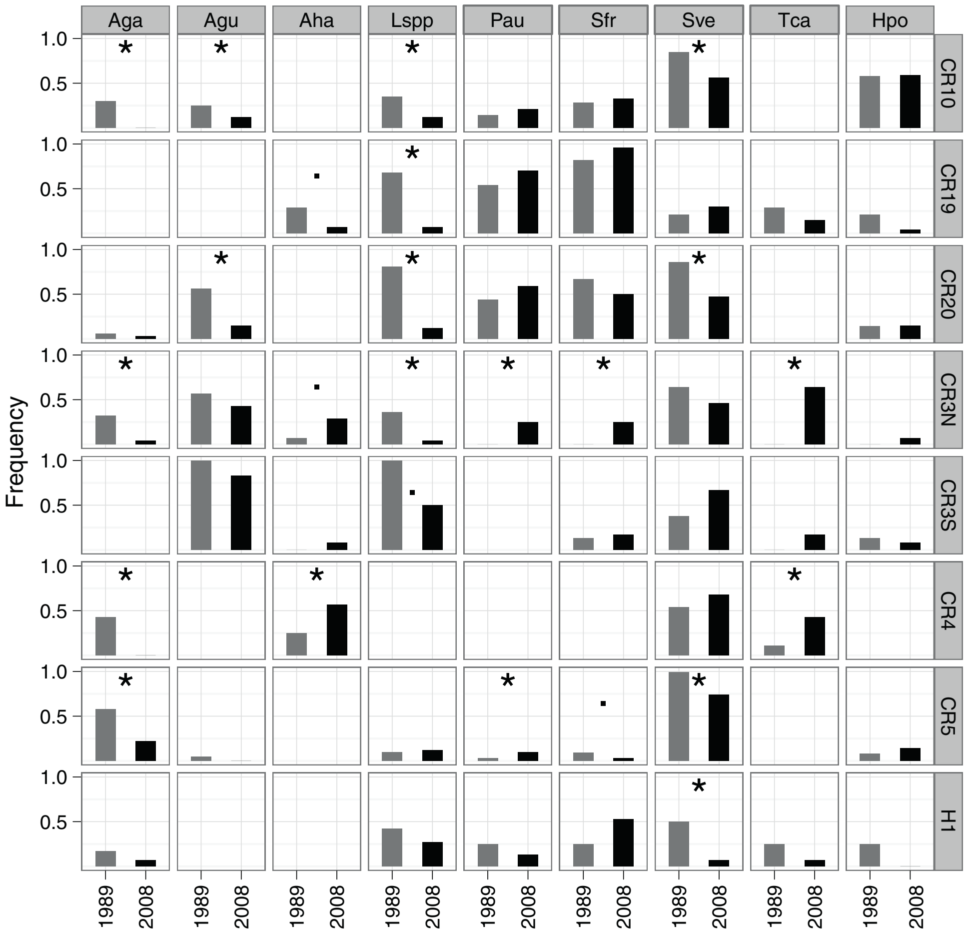

Fig. 3. Frequency of sampled taxa in 1989 and 2008 at each wetland site. Changes at a significance level of $P < 0.1$ after the pairwise comparison binomial test are denoted with a dot, and changes at significance level of $P < 0.05$ with an asterisk. Taxa keys: *Atriplex glauca* (Aga), *A. glaucum* (Agu), *Atriplex halimus* (Aha), *Limonium* spp. (Lspp), *Phragmites australis* (Pau), *Sarcocornia fruticosa* (Sfr), *Suaeda vera* (Sve), *Tamarix canariensis* (Tca) and *Halimione portulacoides* (Hpo). Wetland keys: Cañada Brusca North (CR3N) and South (CR3S), Alcanara (CR5), Marina del Carmolí (CR10), Playa de la Hita (CR20), Matalentisco (CR4), Boquera de Tabala (CR19) and Salinas del Rasall (H1).

which the same taxa and watershed information was recorded and calculated for year 2008. Finally, the resulting index was tested against watershed hydrological condition by means of regression analysis.

3. Results

3.1. Watershed hydrological condition index

Overall, irrigated agricultural land areas increased across wetland watersheds during the study period. While the cover of irrigated land areas (ILA-IDW) represented less than 40% in 1987, it reached values over 60% in 2008 across wetland watersheds (Fig. 2). While Cañada Brusca North watershed showed the highest increase in irrigated land areas during the study period, they were absent in Cañada Brusca South and Salinas del Rasall watersheds during the study period (our reference wetlands). After wetland area relative percentages (WARP) were calculated, larger wetlands showed relatively lower irrigated agricultural land areas values than smaller ones.

3.2. Vegetation changes and selection of indicator taxa

The binomial paired test showed a total of 24 significant changes in plant taxa frequency across wetlands during the study period (Fig. 3). Taxa occurring at a specific wetland with frequencies lower than 0.05 in both years were disregarded from further analyses.

Cañada Brusca North underwent the highest number of significant changes in taxa frequency, while Cañada Brusca South and Salinas del Rasall wetlands (our reference sites) showed very few significant changes in taxa frequency during the study period. Across wetlands, a decreasing trend in frequencies of *Limonium* spp., *A. glaucum* and *Atriplex glauca* was observed (Fig. 3). On the contrary, frequencies of *P. australis*, *T. canariensis* and *A. halimus* generally increased in the study wetlands. Frequencies of *Suaeda vera* and *Sarcocornia fruticosa* did not show a consistent pattern of changes across sites. *Halimione portulacoides* showed no significant frequency changes during the study period and hence it was disregarded as a potential indicator taxon.

Final indicator taxa of watershed agricultural pressures were identified by means of regression analysis. Five out of the eight potential indicator taxa showed significant relationships with weighted land cover percentages (Fig. 4). While *A. glaucum* frequency was positively related to natural areas in watersheds in 1989 and 2008 (Fig. 4A and B), observed frequency changes in *A. halimus* and *T. canariensis* were inversely correlated with changes in natural areas during the study period (Fig. 4D and F). On the other hand, *Limonium* spp. showed a negative relationship with irrigated land areas in 2008 (Fig. 4C), while *P. australis* frequency changes were positively related to changes in irrigated land areas during the study period (Fig. 4E).

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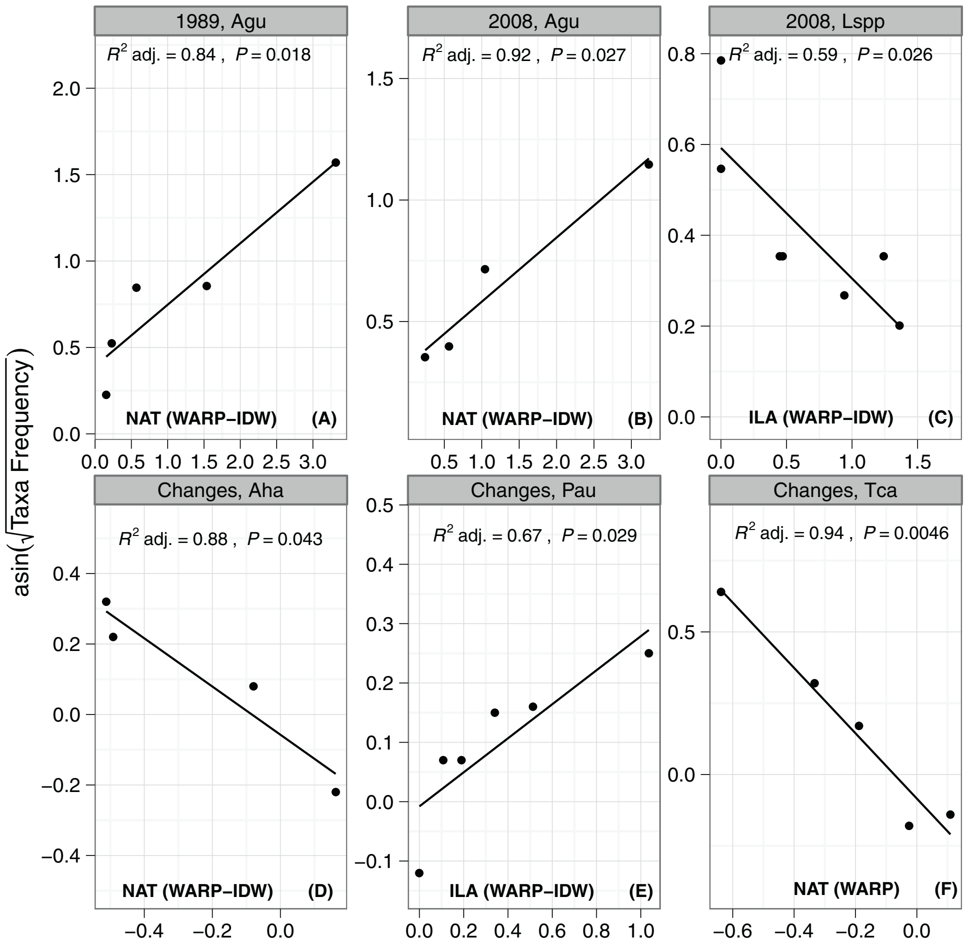

Fig. 4. Significant linear relationships between taxa frequency and weighted land cover percentages (natural and irrigated land areas) in 1989, 2008 and observed changes: (A) and (B) *Arthrocnemum glaucum* (Agu), (C) *Limonium* spp. (Lspp), (D) *Atriplex halimus* (Aha), (E) *Phragmites australis* (Pau), and (F) *Tamarix canariensis* (Tca).

3.3. Wetland condition index

The developed wetland condition index was finally proposed as follows (Eq. 3):

$$WCI = \sqrt{Lspp + Agu} - (Pau + Aha + Tca)^2 \tag{3}$$

where *Lspp* stands for frequency of *Limonium* spp., *Agu* for *A. glaucum*, *Pau* for *P. australis*, *Aha* for *A. halimus* and *Tca* for *T. canariensis*. The wetland condition index ranged from −9 up to 1.4 (Fig. 5), standing negative values for highly altered wetlands. The proposed WCI showed the highest values at the reference wetlands in 2008 (Salinas del Rasall and Cañada Brusca South), while Cañada Brusca North showed the highest decrease during the study period. Overall, except for for Salinas del Rasall wetland, higher values of WCI were observed in 1989 than in 2008.

In 2008, wetland condition index values, including the additional sites, were negatively related to irrigated land areas in watersheds (WARP-IDW) and to observed changes during the study period (Fig. 6A and C). The resulting linear model for 2008 was robust, despite of including extreme values of high watershed hydrological pressures and low wetland condition index, as is the case for Sombrerico wetland (Fig. 6B).

4. Discussion

The proposed wetland condition index (WCI) allows the assessment of Mediterranean semiarid saline wetlands located in agricultural catchments. Inverse distance weighting (IDW) and wetland area relative percentage index (WARP) were useful weighting factors for the assessment of watershed condition in relation to agricultural hydrological pressures on wetlands. Accurate and medium resolution assessment of land cover types and the delimitation of specific watershed areas for each wetland was highly important, as it has been pointed out in previous studies (McHugh et al., 2007; Roth et al., 1996). This was specially important for some complex wetlands like Cañada Brusca, whose fragments (North and South) receive clearly different surface hydrological influences. Moreover, our study revealed that these wetlands showed the most contrasting results in terms of taxa changes during the study period (Fig. 3).

Salinas del Rasall was the only wetland in which WCI increased during the study period, which might be related to the fact that it was legally protected for four years before our study started (Fig. 5). On the contrary, WCI decreased in Cañada Brusca South during the study period, probably due to marginal influences coming from the nearby located watershed of Cañada Brusca North, which suffered the highest ILA increase, followed by Matalentisco wetland (Fig. 2). The fact that Matalentisco and Boquera de Tabala wetlands showed negative WCI values in 1989, might be related to their lack of protection status during the study period. The proposed wetland condition index showed a robust relationship with ILA (WARP-IDW) values in 2008 on an extended set of wetlands (Fig. 6B). Observed changes in WCI and in ILA (WARP-IDW) in their watersheds during the study period were also significantly related (Fig. 6C).

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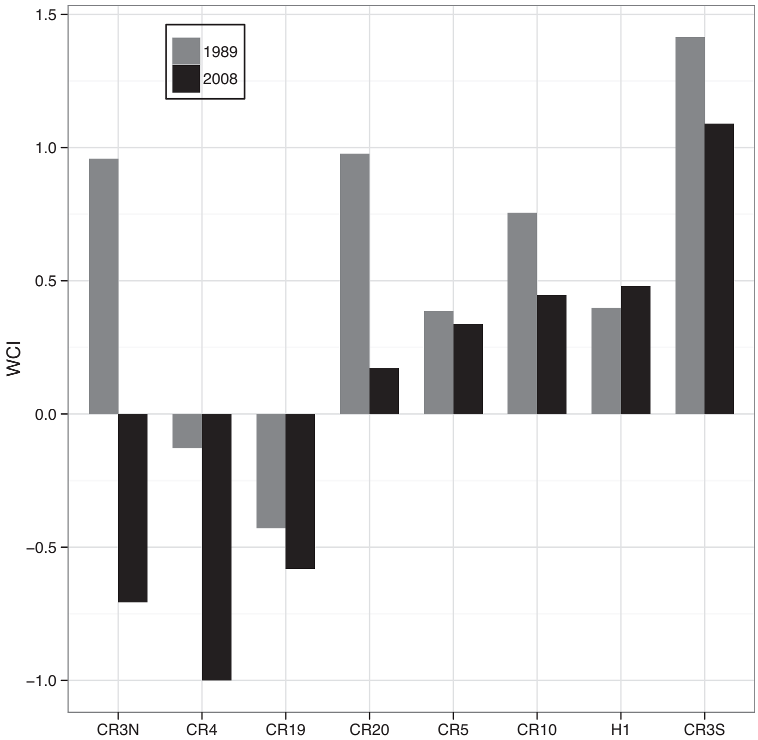

Fig. 5. Wetland condition index (WCI) values for each wetland in 1989 and 2008. Wetlands are arranged from left to right according to a decreasing gradient of irrigated agricultural land areas changes in their watersheds during the study period. Wetland keys: Cañada Brusca North (CR3N) and South (CR3S), Alcanara (CR5), Marina del Carmolí (CR10), Playa de la Hita (CR20), Matalentisco (CR4), Boquera de Tabala (CR19) and Salinas del Rasall (H1).

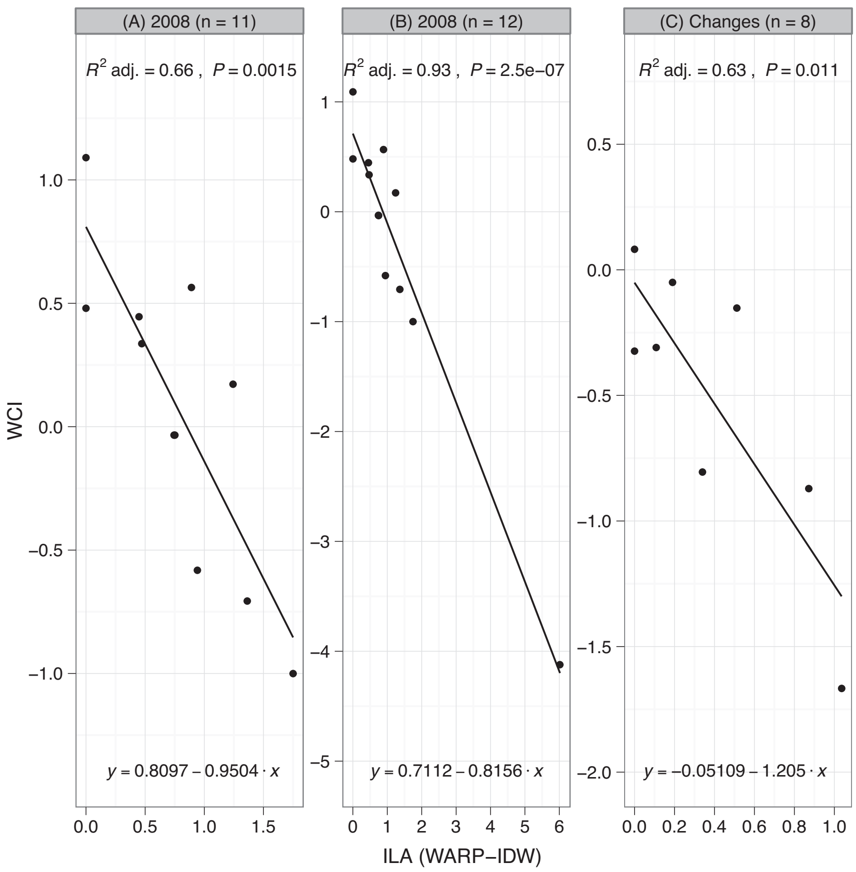

Fig. 6. Linear regression between the wetland condition index (WCI) and irrigated land areas (ILA WARP-IDW) values. Panels A and B show the resulting models in 2008 excluding (n = 11) and including (n = 12) Sombrerico wetland, respectively. Panel C shows the resulting model based on observed changes during the study period (n = 8).

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Historical information reflecting ecosystem responses to land cover changes enhanced the indicator selection process. Undisturbed or reference sites (present or past) were needed in order to establish reference conditions, despite wetlands natural variability. Although our study was constrained by the number of wetlands sampled in 1989, it was based on a long-term period of time and on a set of heterogeneous wetlands, embedded in a representative range of hydrologically altered watersheds. In 1989, irrigated land areas were not the dominant land cover types (Fig. 2), therefore we expected lower influence of watershed land cover on wetland plant taxa at that time. On the contrary, the land use intensification gradient across watersheds was large enough to show relationships with some wetland plants in 2008. Moreover, the analysis of land cover and taxa frequency changes revealed which taxa benefited from the expansion of irrigated land areas during the study period.

The expansion of irrigated land areas at watershed scale seems to have altered wetlands natural hydrological regimes, probably in terms of changes in salinity, flooding frequency and nutrient inputs from agricultural runoff (Robledano et al., 2010). Salinity conditions and seasonal hydrological fluctuations might have decreased across the study wetlands, thus negatively affecting taxa which are adapted to the natural conditions of these wetland types and promoting some taxa adapted to less saline conditions and/or longer flooding periods.

As it has been pointed out by previous studies, *A. glaucum* is known to require saline and humid conditions, being adapted to seasonal fluctuations of water table levels (Álvarez-Rogel et al., 2006; Caballero, 1999; Pujol Fructuoso, 2002). It was the only taxon showing significant relationship with watershed land cover in 1989 (Fig. 4A), thus probably serving as an early warning indicator of the presence of irrigated land areas in the watershed. *Limonium* spp. belongs to the drier part of the humidity gradient, avoids flooded soils and withstands high salinities (Caballero, 1999; Álvarez Rogel et al., 2000). *P. australis* is considered an invasive species (Álvarez-Rogel et al., 2007; Tulbure et al., 2007) usually forming monospecific stands, reducing species diversity, and therefore indicating wetland degradation. Its frequency increased in most wetlands along with higher percentages of irrigated land areas in the watershed (Fig. 4E), probably as a consequence of lower salinity and longer flooding periods (Burdick and Konisky, 2003). *A. halimus* occurrence is associated with lower salinity conditions than salt steppe and salt marsh communities in the drier part of the humidity gradient and might be found in association with salt cedar patches (MARM, 2009).

5. Conclusions

By mean of historical fieldwork sampling, remote sensing and hydrological modeling techniques, a wetland condition index (WCI) was developed, based on plant taxa frequency, that allows the assessment of Mediterranean semiarid saline wetlands located in agricultural catchments, serving as a management tool to preserve their values and associated ecosystem services. The results from this study reinforce the importance of quantifying the influence of landscape level processes for wetland management and conservation, in accordance with previous studies (Rooney et al., 2012). International management efforts for protecting freshwater ecosystems in Mediterranean areas, like the European Water Framework Directive (European Commission, 2000), the European Habitats Directive (Council of Europe, 1992) and the RAMSAR convention (Ramsar Convention Secretariat, 2004; Finlayson, 2005), should specifically take into account catchment-scale hydrological influences of agricultural land uses on wetlands.

Acknowledgements

Financial support was provided by the project "Estado Ecológico de los Humedales Mediterráneos Semiáridos: Propuesta de Indicadores para su Evaluación" (I+D No: CGL2006-08134) and an associated FPI grant for J. Martínez-López. The authors acknowledge the comments from María Almagro and two anonymous reviewers on an earlier version of the manuscript.

Appendix A. Supplementary data

Supplementary data associated with this article can be found, in the online version, at http://dx.doi.org/10.1016/j.ecolind.2013.08.007.

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