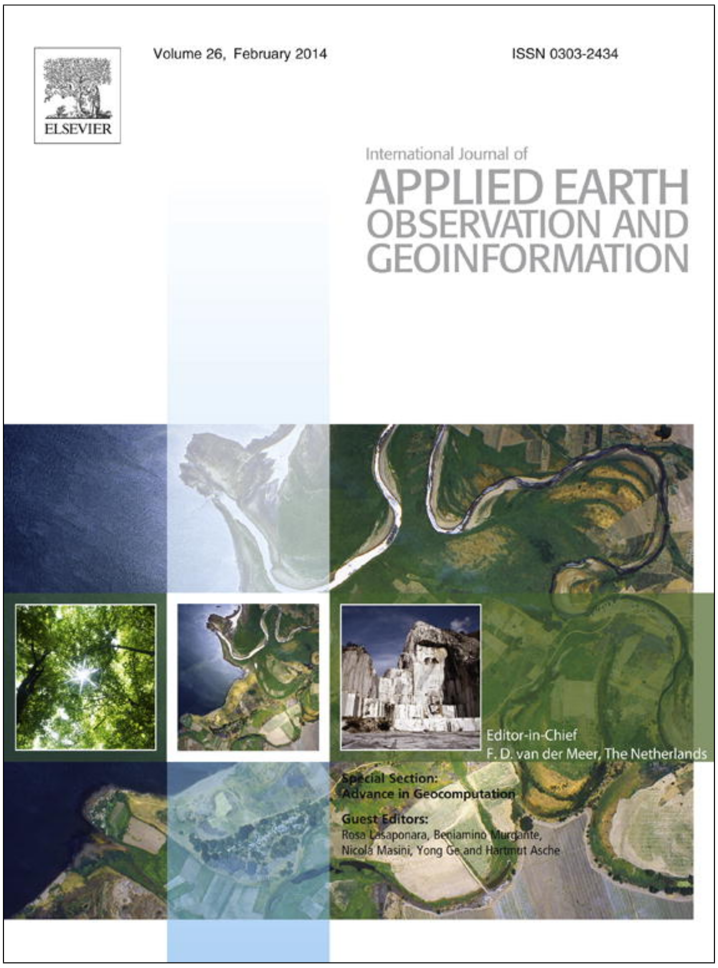
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Remote sensing of plant communities as a tool for assessing the condition of semiarid Mediterranean saline wetlands in agricultural catchments

J. Martínez-López*, M.F. Carreño, J.A. Palazón-Ferrando, J. Martínez-Fernández, M.A. Esteve

Ecology and Hydrology Department, Faculty of Biology, Campus de Espinardo, E-30100 Murcia, Spain

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ABSTRACT

Semiarid Mediterranean saline wetlands are unique ecosystems sheltering high biodiversity. In the last decades, the expansion of irrigated lands has led to hydrological imbalances in Mediterranean catchments, causing wetland degradation. Vegetation composition assessment is considered an important tool for evaluating wetland ecological condition and can be mapped using remote sensing. This study aims to develop a condition index based on plant community composition suitable for semiarid Mediterranean saline wetlands, as well as to test the applicability of airborne multispectral remote sensors for discriminating plant communities. Characteristic plant communities of 12 wetlands were identified by means of ordination and classification analysis of plant taxa cover percentages obtained through fieldwork sampling. An index for assessing wetland ecological condition was developed based on the relationship between wetland plant community composition and watershed hydrological condition. Selected wetland plant communities were then mapped by means of remote sensing techniques using random forest algorithm for supervised classification of airborne images. Following this methodology, remote sensing served as a tool for wetland condition assessment at a regional scale.

1. Introduction

Semiarid Mediterranean saline wetlands are semi-terrestrial ecosystems, which yearly undergo dry periods of several months, and shelter a rich, endemic and sensitive biota (Brinson and Malvárez, 2002; Vidal-Abarca et al., 2003). They are sinks of irrigation flows, which makes catchment scale management of vital importance for their conservation (Dudgeon et al., 2006). In the last decades, the expansion of irrigated areas in semiarid Mediterranean catchments has led to altered inputs of water and nutrients to lowland wetlands (Hollis, 1990; Carre no et al., 2008; Martín-Queller et al., 2010), which particularly affect soil salinity, water table level and regime and soil moisture conditions (Álvarez-Rogel et al., 2007b). Although much effort has been applied towards protection of wetlands, the preservation of their watershed areas has been largely ignored (Houlahan and Findlay, 2004). Moreover, the lack of systematic monitoring and management procedures for this type of saline aquatic ecosystems is ultimately leading to extensive wetland degradation, and therefore practical tools for assessing ecosystem state and functioning are required in order to orient decision making (Williams, 2002).

Vegetation composition assessment is considered an important aspect for evaluating wetland ecological condition (López and Fennessy, 2002; Miller et al., 2006; García et al., 2009; Caçador et al., 2013). Anthropic or climatic factors that affect wetland plant communities also affect wetland birds and invertebrates communities (Hughes, 2004; Pardo et al., 2008; Robledano et al., 2010). The European Habitats Directive considers salt marsh and salt steppe habitats as important and endangered ones and promotes their preservation (Council of Europe, 1992). Moreover, the European Water Framework Directive also impels to monitor the ecological status of transitional waters ecosystems including wetlands (European Commission, 2003; Ferreira et al., 2007). Plant species and communities can be used as a proxy to assess wetland hydrological perturbations if their ecological tolerances to environmental factors such as salinity and water table level are known (Cronk and Fennessy, 2001). Previous studies in similar wetlands have focused on individual taxa rather than on plant communities (Álvarez-Rogel et al., 2007b; Martínez-López et al., 2012). However, plant communities contain more information than single species and are easier to map by means of remote sensors (O'Connell, 2003; Johnston et al., 2009).

* Corresponding author. Tel.: +34 868884653; fax: +34 868883963.
E-mail addresses: javier.martinez@um.es, javi.martinez.lopez@gmail.com (J. Martínez-López), juliamf@um.es (J. Martínez-Fernández).

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Given that wetlands are often located in remote areas and are difficult to survey, fieldwork sampling methods are labour intensive, specially for large wetland sites and for regional scale assessment. Therefore, most of these ecosystems receive no systematic monitoring (Buchanan et al., 2009). In this regard, remote sensing offers a rapid and cost-effective approach for generating ecologically relevant wetland vegetation maps (Jones et al., 2009; Lee and Yeh, 2009; Poulin et al., 2010). Remote sensing techniques are able to cover large areas, are prone to rapid technical improvements, and can help discriminate different wetland plant communities in order to systematically assess wetland conservation status (Belluco et al., 2006; Wang et al., 2007). Due to pixel size, specific plant species are usually more difficult to discriminate, unless they form dense monospecific stands (*e.g.* reed beds). However, plant communities are composed by similar relative proportions of specific plant species across wetland sites and therefore they tend to show characteristic spectral responses at large patches. Therefore, the establishment of plant communities that can be related to wetland hydrological pressures combined with remote sensing techniques can serve as a tool for wetland management and monitoring at a regional scale (Ozesmi and Bauer, 2002; Xie et al., 2008).

Different methods have been developed for wetland vegetation delimitation from remote sensing imagery (Horning et al., 2010; Friess et al., 2012; Szantoi et al., 2013). Wetlands are very heterogeneous ecosystems containing small vegetation patches with similar spectral responses that might generate spectral confusion (Ozesmi and Bauer, 2002; Baker et al., 2006). Enhanced classification methods like random forest algorithm or artificial neural networks have shown better performance for these Mediterranean ecosystems (Breiman, 2001; Černá and Chytrý, 2005; Sluiter, 2005). Medium-resolution satellite sensors such as Landsat TM are not suitable for detecting small wetlands or plant community patches (Xie et al., 2008; Adam et al., 2010). Therefore, airborne multispectral sensors are a good source of remotely sensed data for wetland vegetation mapping since they combine high spatial and spectral resolution with acquisition timing (Wang et al., 2007; Klemas, 2011).

The scarcity of available historical data on wetland vegetation due to inaccessibility and high economic cost calls for the use of remote sensing for present and future studies in order to provide a set of multi-temporal images to monitor wetland ecosystems (Klemas, 2001; Carre no et al., 2008; MacKay et al., 2009). Scientific studies bridging the gap between ecology and conservation biology are utmost important in order to influence conservation of aquatic ecosystems (Strayer and Dudgeon, 2010). This study proposes a procedure for a comprehensive study of wetlands and their drainage basins coupling fieldwork and advanced remote sensing techniques in order to evaluate wetland condition based on plant communities. The main objectives of this study were: (1) to characterize plant communities in several representative wetlands under a range of watershed hydrological conditions; (2) to explore the relationships between wetland plant community composition and watershed hydrological pressures; (3) to propose a wetland condition index based on plant community composition; and (4) to test the potential of remote sensing as a tool for assessing wetland condition.

2. Methods

2.1. Study wetlands

The study area was located in Murcia province, southeast of Spain (38°45′–37°23′ North and 0°41′–2°21′ West). The climate is semiarid Mediterranean, with a mean annual temperature of 16 °C and a mean annual precipitation of 339 mm (Esteve et al., 2006).

Table 1
Study wetlands and respective watershed areas (ha).

Wetland	Wetland area	Watershed area
Salinas del Rasall	26.3	236
Saladar de Cañada Brusca South	3.8	69.5
Saladar de Cañada Brusca North	17.4	360
La Alcanara	199	6508
Marina del Carmolí	314	16,923
Playa de la Hita	34.4	2052.8
Saladar de Matalentisco	10.4	907.6
Saladar de Boquera de Tabala	36.9	5819.2
Lopoyo	80	2783
Ajauque	100	7792
Derramadores	50	1963
Sombrerico	3	141

Twelve representative wetlands were selected, *i.e.* 8 coastal and 4 inland wetlands (Figs. 1 and 2; 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 rules 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, Lo Poyo 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 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

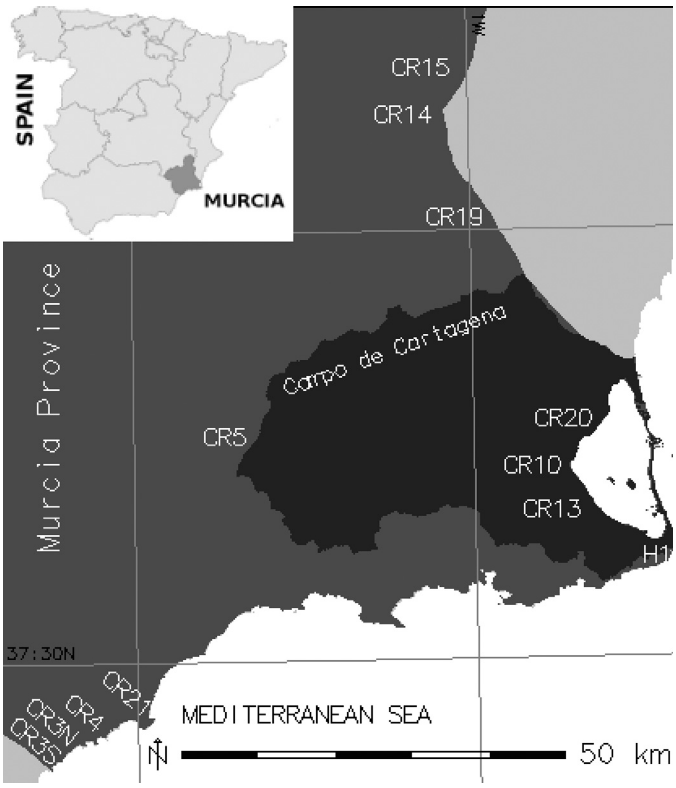

Fig. 1. Location of Murcia province in Spain and approximate location of the study wetlands. Wetland keys: H1: Rasall; CR3S: Cañada Brusca South; CR10: Carmoli; CR5: Alcanara; CR14: Ajauque; CR13: Lopoyo; CR15: Derramadores; CR19: Boquera de Tabala; CR20: Playa de la Hita; CR3N: Cañada Brusca North; CR4: Matalentisco; CR21: Sombrerico.

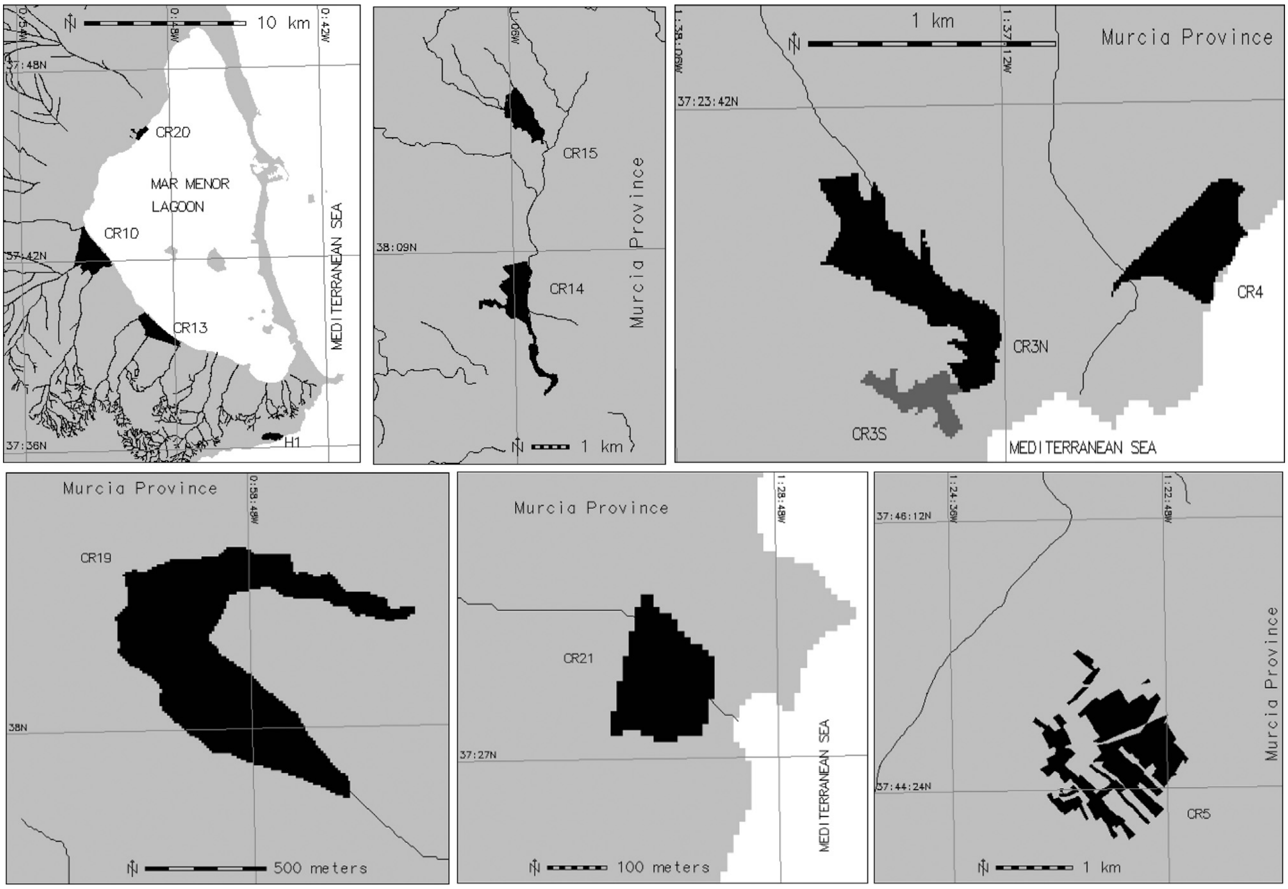

Fig. 2. Wetlands area maps and relation to the river network. Wetland keys—H1: Rasall; CR3S: Cañada Brusca South; CR10: Carmoli; CR5: Alcanara; CR14: Ajauque; CR13: Lopoyo; CR15: Derramadores; CR19: Boquera de Tabala; CR20: Playa de la Hita; CR3N: Cañada Brusca North; CR4: Matalentisco; CR21: Sombrerico.

Park, and also included in the Mar Menor RAMSAR protected area. Matalentisco, Sombrerico and Cañada Brusca North and South are coastal wetlands located in the southern part of the region on the Mediterranean Sea. Boquera de Tabala, Ajauque, Derramadores and Alcanara are inland wetlands associated with an ephemeral river and with a saline alluvial plain, respectively.

Plant communities of these wetlands are salt steppes and salt marshes, which occupy areas with high soil salinity and intermittent waterlogging. Salt steppe is 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. Salt marsh is dominated by habitat *1420* (Mediterranean and thermo-Atlantic halophilous scrubs) and habitat *1410* (Mediterranean salt meadows). Main species in salt marsh are *Sarcocornia fruticosa*, *Arthrocnemum macrostachyum*, *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*.

2.2. Wetland basin characteristics

Specific watershed boundaries for each wetland were delineated from a raster DEM with a pixel size of 10 m using single flow direction method (D8 algorithm). Prior to watershed delineation, the digital elevation model (DEM) was modified in the Campo de Cartagena coastal plain area to enhance the delineation of wetland watersheds. To accomplish this, elevation values coinciding with existent stream network were lowered to force flow-direction models to match existing stream lines (Strayer et al., 2003; King et al., 2005). 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 also 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).

Since irrigation flows from near irrigated areas were supposed to exert more influence on wetland hydrology (Casta neda and Herrero, 2008) and their effect on larger wetlands should be considerably lower than in smaller ones, raw percentages of irrigated areas in the watersheds were weighted by landscape metrics, such as inverse-distance weights (King et al., 2005; Van Sickle and Johnson, 2008) and receiving wetland area. A wetland area relative percentage index (WARP) was thus developed in order to assess and compare the hydrological pressures that irrigation at watershed scale exerts on different wetlands (Eq. (1)):

$$\mathrm{WARP_{ILA}} = \sqrt{\frac{\mathrm{ILA\,(IDW)\ in\ watershed\ (\%)}}{\mathrm{Wetland\ area\ (ha)}}} \tag{1}$$

where ILA (IDW) refers to raw percentages of irrigated land areas in the watersheds, which were first inverse-distance weighted (IDW).

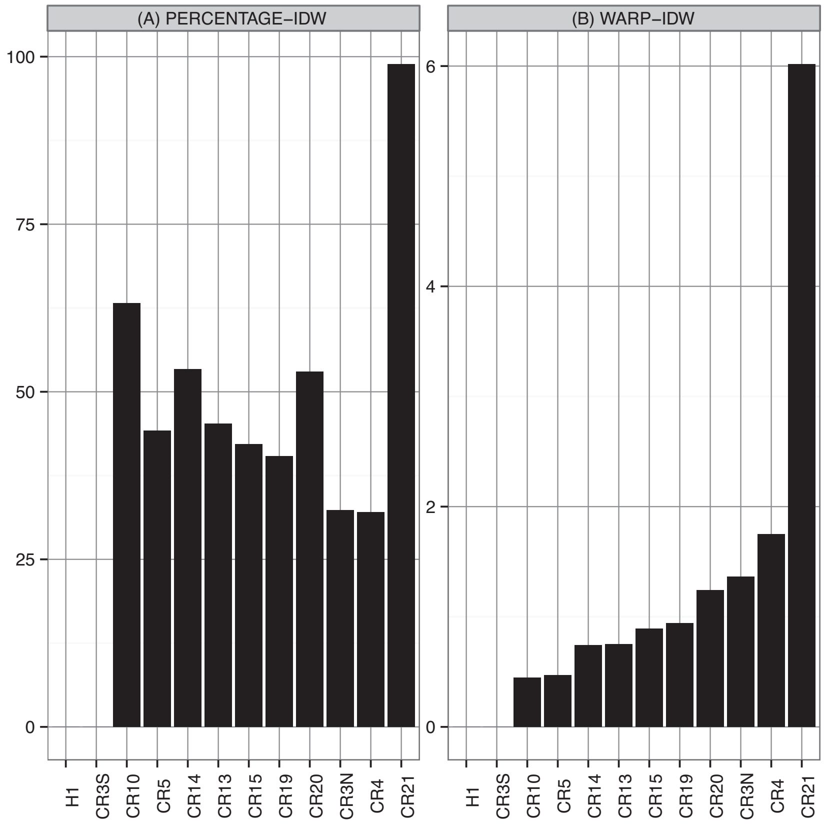

Fig. 3. Irrigated agricultural land areas (ILA) land cover percentages (inverse distance weighted: IDW) (A) and wetland area relative percentages (WARP-IDW) (B) in wetland watersheds in 2008. Wetland keys—H1: Rasall; CR3S: Cañada Brusca South; CR10: Carmoli; CR5: Alcanara; CR14: Ajauque; CR13: Lopoyo; CR15: Derramadores; CR19: Boquera de Tabala; CR20: Playa de la Hita; CR3N: Cañada Brusca North; CR4: Matalentisco; CR21: Sombrerico.

Maps of irrigated areas from each wetland watersheds in 2008 were obtained from previous studies, where land cover maps were produced by means of supervised classification of Landsat images at 25 m resolution (Martínez-López et al., 2012).

2.3. Vegetation field sampling

Across wetlands, a total of 1843 georeferenced sampling plots (2 m × 2 m) were surveyed between April and June 2009. The number of vegetation sampling plots within each wetland ranged from 30 to 550, systematically located at different intervals depending on wetland size and heterogeneity of plant communities. At each sampling plot, selected plant taxa cover was recorded (0–100%), together with upland species and bare soil cover. According to known qualitative ranges of tolerance to salinity and waterlogging, 21 representative perennial taxa were sampled: *Suaeda vera* Forssk. ex J.F. Gmel., *Atriplex halimus* L., *Atriplex glauca* L., *Arthrocnemum macrostachyum* (Moric.) Moris in Moris & Delponte, *Halimione portulacoides* (L.) Aellen, *Halocnemum strobilaceum* (Pall.) M. Bieb., *Sarcocornia fruticosa* (L.) A.J. Scott, *Suaeda pruinosa* Lange, *Scirpus holoschoenus* L., *Frankenia corymbosa* Desf., *Phragmites australis* (Cav.) Trin. ex Steud., *Lygeum spartum* Loefl. ex L., *Imperata cylindrica* (L.) Raeusch., *Juncus* sp. L., *Limonium caesium* (Girard.) Kuntze, *Limonium cossonianum* Kuntze, *Plantago crassifolia* Forssk., *Sarcocornia perennis* (Mill.) A.J. Scott, *Saccharum ravennae* (L.) Murray, *Tamarix boveana* Bunge and *Tamarix canariensis* Willd. Several sampling plots were disregarded due to the presence of infrastructures or other disturbances.

2.4. Ordination and classification analysis

Vegetation sampling data were grouped using ordination and classification analysis to identify vegetation clusters which might be related to the ones described by the European Habitat Directive. Taxa that are present in less than 1% of the sampling plots were discarded from the analysis. The final data matrix contained 13 taxa and 1684 sampling plots. The taxa-abundance matrix was first transformed into a dissimilarity matrix using the Bray–Curtis dissimilarity index. Hierarchical clustering was then performed and the resulting classification tree was cut in groups, representing plant communities. In order to determine significant indicator taxa of each plant community type indicator value analysis (IndVal) was applied (Roberts, 2010). Non-metric multi dimensional scaling (MDS) was also conducted (Oksanen et al., 2011) to graphically represent and assess the fit of the ordination analysis. SIMPER procedure (Clarke, 1993) was used to determine average dissimilarities between vegetation clusters. Finally, analysis of similarities (ANOSIM) (Clarke, 1993) was carried out to test whether there were significant differences in plant composition between the obtained communities. All statistical analyses were conducted using R (R Core Team, 2012).

2.5. Wetland ecological condition index based on plant community composition

Supported by the knowledge on the ecology of each studied plant community type, and its sensitivity to hydrological related factors, a wetland condition index was sought, that could

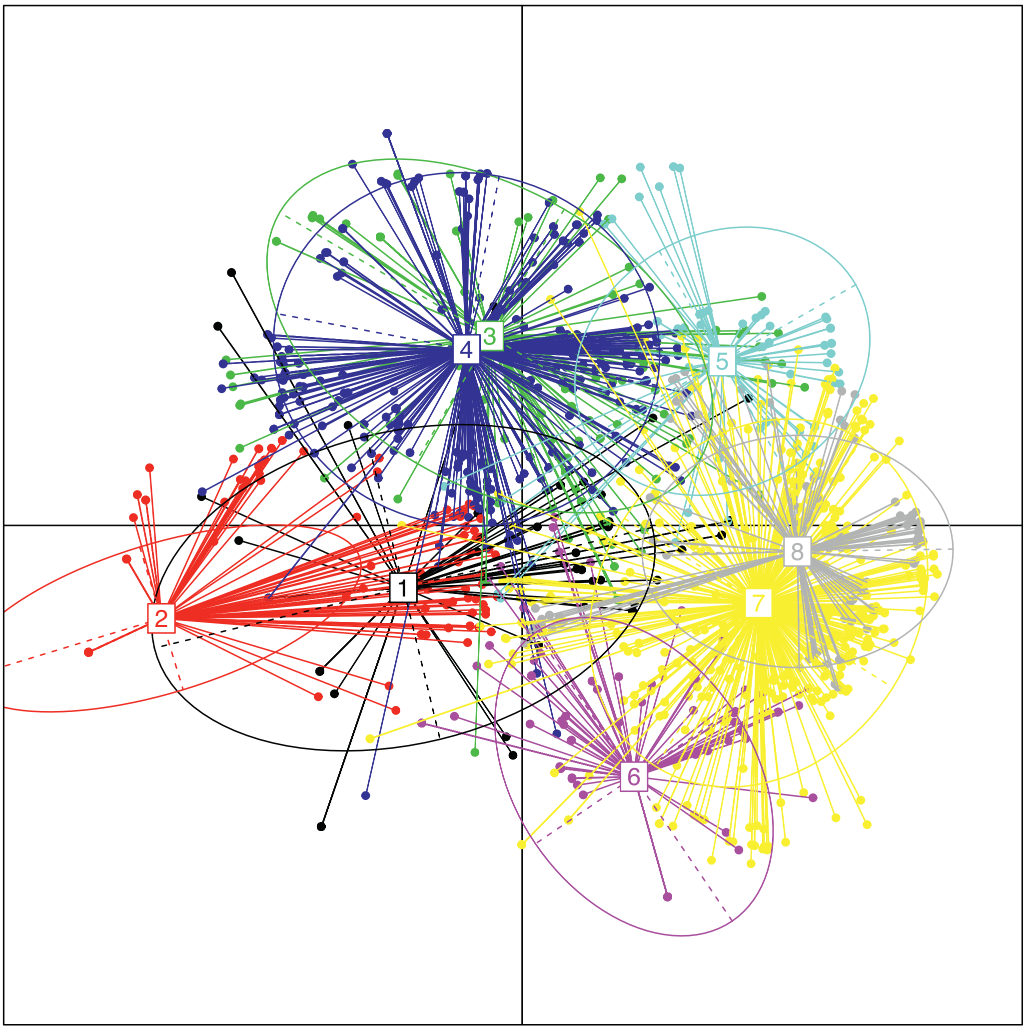

Fig. 4. Non-metric multidimensional scale graph. Plant community keys—1: *Tamarix* sp. and *Atriplex halimus* mixed community (Tca-Aha); 2: *Phragmites australis* dominated community (Pau); 3: salt marsh dominated by *Arthrocnemum macrostachyum* (SM-Ama); 4: salt marsh dominated by *Sarcocornia fruticosa* (SM-Sfr); 5: salt marsh dominated by *Halimione portulacoides* (SM-Hpo); 6: dominated by upland plants (UP); 7: salt steppe (SS); 8: *Suaeda vera* dominated community (Sve).

serve as a measure of semiarid Mediterranean saline wetland ecological condition status in relation to watershed agricultural hydrological pressures. In order to quantitatively analyze wetland plant communities composition and summarize them into a meaningful biological gradient, correspondence analysis (CA) was performed on the resulting wetland communities-abundance matrix (Benzécri, 1973; Husson et al., 2010). Hydro-ecological linkages between wetland plant communities composition (CA axes) and watershed hydrological condition (ILA WARP-IDW) were then established by means of regression analysis, and thus an index of wetland condition (WCI_{PC}) based on plant community composition was developed.

2.6. Plant communities mapping

Plant communities of the studied wetlands were mapped by means of supervised classification of an image with a pixel size of 2 m, obtained with an airborne multispectral sensor in 2008 (DGPNB, 2008). The sensor used was a *DMC Z/I Intergraph* camera with four spectral bands, *i.e.* red (R, 590–675 nm), green (G, 500–650 nm), blue (B, 400–580 nm) and near infrared (NIR, 675–850 nm). Normalised difference vegetation index (NDVI; Rouse et al., 1973) was calculated and was included as an ancillary layer in classification analysis. Since the image corresponding to Playa de la Hita (CR20) wetland was damaged, plant communities from this site could not be mapped.

The classification procedure consisted of two steps. Firstly, unsupervised classification of 20 spectral clusters was performed to discriminate readily identifiable non-vegetation cover classes. According to photointerpretation using an aerial photography obtained from the same flight (0.45 m pixel size), the obtained classes were identified as infrastructures, water bodies and open spaces of bare soil and were masked out from further classification analysis. Secondly, supervised classification of image was performed in order to discriminate wetland plant communities. For this purpose, georeferenced sampling plots (2 m × 2 m) surveyed in the field were used as training and validation areas for image classification. Fifty percent of the pixels belonging to each plant community type were randomly selected and assigned to the training and validation maps, respectively. Although airborne image dated from June 2008, almost one year earlier than field sampling took place, plant species composition is known to be relatively constant in time in these ecosystems (Zedler et al., 1999) and almost no vegetation changes were expected during this period since no specific disturbances occurred. Random forest (Breiman, 2001) was the algorithm used for supervised classification. This non-parametric classifier grows a set of 500 decision trees using a different subset of train areas each time, allowing to vote for the most popular

Table 2
Indicator species for each vegetation cluster after the IndVal Analysis. Associated probability value and corresponding code of the EU Habitats Directive is indicated.

Cluster name	Species	Indicator value	Probability	EU Habitat
SM-Ama	*A. macrostachyum*	0.786	0.001	1420
SM-Sfr	*S. fruticosa*	0.8148	0.001	1420
	Juncus sp.	0.0888	0.001	
SM-Hpo	*H. portulacoides*	0.7916	0.001	1420
SS	*F. corymbosa*	0.3618	0.001	1510
	L. spartum	0.3183	0.001	
	L. caesium	0.0961	0.001	
	L. cossonianum	0.0566	0.002	
Pau	*P. australis*	0.8596	0.001	–
Tca-Aha	*Tamarix* sp.	0.6983	0.001	92D0/1430
	A. halimus	0.4688	0.001	
Sve	*S. vera*	0.5408	0.001	1430
	A. glauca	0.0869	0.001	
UP	*Upland plants*	0.7747	0.001	–

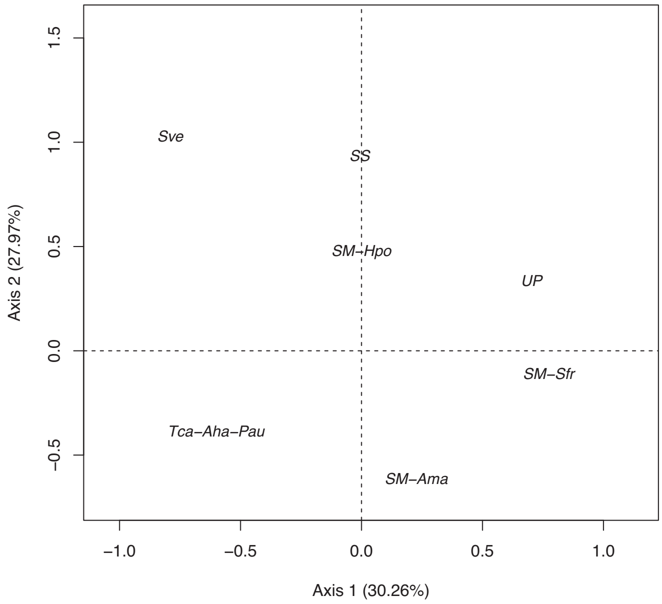

Fig. 5. Correspondence analysis (CA) of wetland plant communities abundance. Plant community keys: Tca-Aha-Pau: *Tamarix* sp. + *Atriplex halimus* + *Phragmites australis*; SM-Ama: salt marsh dominated by *Arthrocnemum macrostachyum*; SM-Sfr: salt marsh dominated by *Sarcocornia fruticosa*; SM-Hpo: salt marsh dominated by *Halimione portulacoides*; SS: salt steppe; Sve: *Suaeda vera*; UP: upland plants.

assigned class, and thus producing a significant increase in classification accuracy (Liaw and Wiener, 2002; Pal, 2005). Classification results were validated by means of the *overall accuracy* parameter and the Kappa coefficient (Foody, 2002). Producer's and user's accuracy parameters were also calculated for each plant community of interest across wetland sites (Congalton, 1991).

3. Results

3.1. Watershed hydrological conditions

The size of wetland watersheds ranged from 70 to 17,000 ha. Percentages of irrigated areas among wetland watersheds ranged from 0% to 99% (Fig. 3A). While Sombrerico watershed showed the highest percentage of irrigated area, they were absent in Cañada Brusca South and Rasall watersheds. When wetland area relative percentages (WARP) were calculated, larger wetlands showed relatively lower agricultural hydrological pressures than smaller ones, and as a result a gradient of watershed hydrological conditions was observed across wetlands (Fig. 3B).

3.2. Plant communities characterization

The resulting hierarchical classification tree was cut in eight clusters and significant indicator taxa of each corresponding plant community type or cluster were identified using the IndVal analysis (Table 2). MDS representation of field sampling plots (stress of 0.18; Fig. 4) and ANOSIM analysis showed that vegetation clusters were clearly distinct ($R^2 = 0.83$; $P = 0.001$). Average dissimilarities between vegetation clusters were above 80% after the SIMPER analysis, except for two clusters dominated by *Suaeda vera* and several salt steppe taxa, respectively, which only reached 69%.

Most of the vegetation clusters obtained were consistent with plant communities described in the Habitat Directive. Typical salt-marsh communities (EU Habitat *1420* – Mediterranean and thermo-Atlantic halophilous scrubs) were differenciated in three groups, dominated by *Arthrocnemum macrostachyum* (SM-Ama), *Sarcocornia fruticosa* (SM-Sfr) and *Halimnione portulacoides* (SM-Hpo). Salt steppe community (SS) was evenly represented by several characteristic taxa and corresponded to EU Habitat *1510* (Mediterranean salt steppes – Limonietalia). A plant community was represented by *Phragmites australis* (Pau), which is not included in the Habitat Directive. *Tamarix* sp. (mainly *Tamarix canariensis*) and *Atriplex halimus* were grouped together in one plant community (Tca-Aha), representing dense Tamarisk forest patches surrounded by *A. halimus*. This type corresponds with a mixed community of EU habitats *92D0* (Southern riparian galleries and thickets) and *1430* (Halo-nitrophilous scrubs Pegano-Salsoletea). Another plant community was mainly represented by *Suaeda vera* (Sve), which also corresponded to a documented subtype of the EU Habitat *1430* composed by pioneering taxa in recently abandoned arable areas (MARM, 2009). Finally, a heterogeneous vegetation cluster was obtained with no specific indicator taxa, composed by several upland ruderal halo-nitrophilous plant species not specifically belonging to wetlands (UP).

3.3. Wetland ecological condition index based on plant community composition

The use of CA allowed us to summarize wetlands plant community composition into a single multivariate axis that represented a biological condition gradient, which can be related to environmental variables. Plant communities represented by *Phragmites australis, Tamarix canariensis* and *Atriplex halimus* were added up and pooled in a single functional plant community type, since all of them have been previously reported as invasive and related to hydrological perturbations (Burdick and Konisky, 2003; Chambers et al., 2003; Álvarez-Rogel et al., 2006) (see Fig. 5).

The first and second axes of the CA explained 30% and 28% of the variance, respectively. In relation to the first axis, characteristic wetland plant communities were grouped in the near zero and positive part of the axis (SM-Sfr, SM-Ama, SM-Hpo and SS), whose ecological tolerances are similar and known to occur under typical wetland abiotic ranges in terms of salinity, water table level and flooding regimes (Álvarez-Rogel et al., 2000, 2007b). On the contrary, the lowest values of the first axis grouped plant communities previously related to hydrological perturbations (Tca-Aha-Pau), together with a community type which is typical from wetland areas altered by *in situ* factors (Sve) (Caballero, 1999). After linear

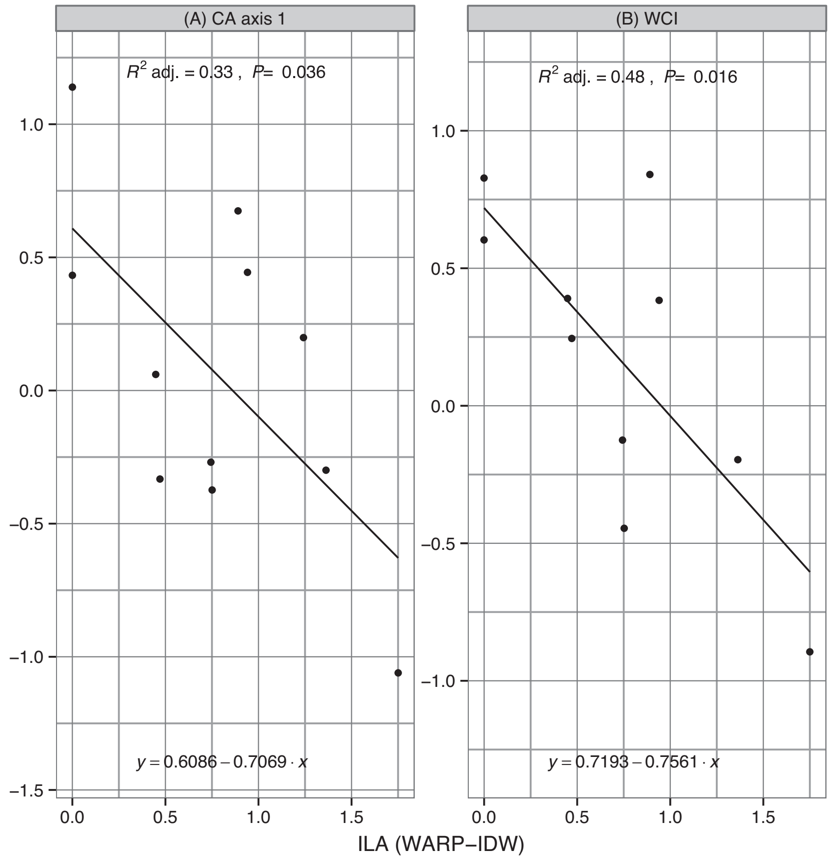

Fig. 6. Relationship between watershed hydrological condition (ILA WARP-IDW) in 2008 and (A) correspondence analysis (CA) axis 1 and (B) plant community based index of wetland condition (WCI_{PC}) as the response variables.

regression analysis, the first axis of the CA was negatively correlated with the irrigated land areas (WARP-IDW) index in 2008 ($R^2_{adj.} = 0.33$; $P = 0.03$) (see Fig. 6A). Sombrerico wetland was considered and outlier and therefore was excluded from the regression analysis since it represented extreme ILA (WARP-IDW) values.

A plant community based index of wetland condition (WCI_{PC}) was thus proposed based on CA axis 1. Relative abundances of plant communities scoring negatively in CA axis 1 were subtracted to abundances of positive scoring communities (see Eq. (2)).

$$WCI_{PC} = (RA_{SM} + RA_{SS}) - (RA_{Tca-Aha-Pau} + RA_{Sve}) \quad (2)$$

where RA_{SM} stands for the sum of the relative abundances of salt marsh communities (SM-Sfr, SM-Ama and SM-Hpo), RA_{SS} stands for relative abundance of the salt steppe community, $RA_{Tca-Aha-Pau}$ stands for the sum of the relative abundances of the Tamarisk and reed beds dominated plant communities, and RA_{Sve} stands for the relative abundance of the *Suaeda vera* dominated plant community, expressed as parts per unit.

The values of the condition index for the study wetlands ranged from −1 to 1 (see Fig. 7). Positive values represent wetland sites with good ecological status, while negative values stand for disturbed sites due to hydrological perturbation. Boquera de Tabala (CR19), Derramadores (CR15) and Cañada Brusca South (CR3S) wetlands scored over 0.5, representing very good condition sites, and Matalentisco (CR4), Sombrerico (CR21) and Lopoyo (CR13) wetlands showed condition values under −0.5, indicating very bad condition.

Interestingly, the proposed plant community based wetland condition index showed a stronger relationship with the irrigated land areas index (WARP-IDW; $R^2_{adj.} = 0.48$; $P = 0.02$; see Fig. 6B) in 2008 than with the first axis of the correspondence analysis. There was no correlation between wetland condition and abundance of upland plants ($R = 0.17$; $P = 0.61$). The presence of this community type is probably related to an inaccurate or old wetland delimitation but no to hydrological pressures coming from the watershed, and therefore we did not include them in the condition index.

3.4. Plant community maps

A separate random forest classification model was performed for wetlands located in the South of Murcia province (CR3N, CR3S, CR4 and CR21) in order to improve classification accuracy at these

Table 3
Overall accuracy and Kappa values for each wetland plant community map. Wetland sites are grouped by the two random forest (RF) models performed (Northern and Southern wetlands). Number of validation areas per wetland is indicated (N). Wetland keys: CR10: Carmoli; CR13: Lopoyo; CR14: Ajauque; CR15: Derramadores; CR19: Boquera de Tabala; CR5: Alcanara; H1: Rasall; CR21: Sombrerico; CR3N: Cañada Brusca North; CR3S: Cañada Brusca South; CR4: Matalentisco.

	Wetland	Overall accuracy	Kappa	*N*
RF North	CR10	74%	0.62	243
	CR13	95%	0.78	42
	CR14	67%	0.55	85
	CR15	72%	0.26	72
	CR19	49%	0.10	51
	CR5	76%	0.51	96
	H1	94%	0.64	16
RF South	CR21	59%	0.20	17
	CR3N	52%	0.06	21
	CR3S	75%	0.33	12
	CR4	50%	0.20	22

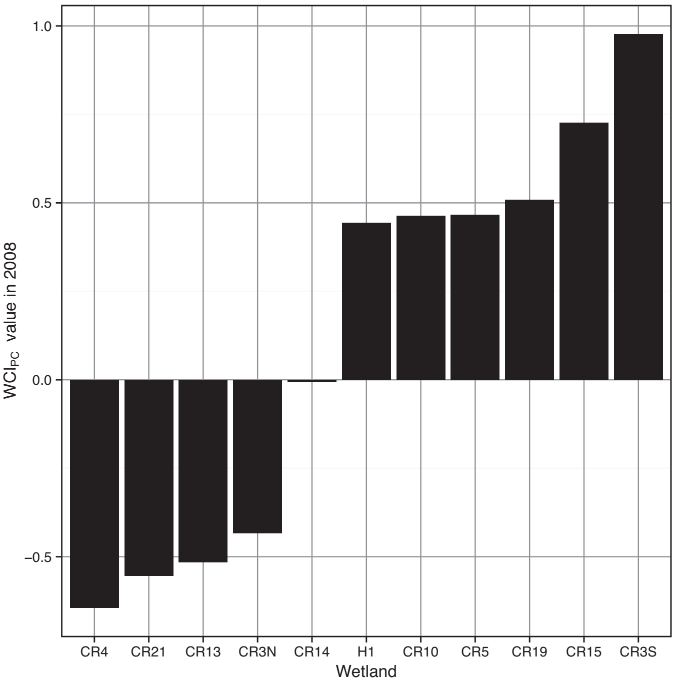

Fig. 7. Values of the plant community based index of wetland condition (WCI_{PC}) in 2008. Wetland keys—CR4: Matalentisco; CR21: Sombrerico; CR13: Lopoyo; CR3N: Cañada Brusca North; CR14: Ajauque; H1: Rasall; CR10: Carmoli; CR5: Alcanara; CR19: Boquera de Tabala; CR15: Derramadores; CR3S: Cañada Brusca South.

sites. All other wetlands were classified using the same random forest model.

Resulting maps were validated for the plant communities included in the wetland condition index, *i.e.* Tca-Aha-Pau, SM (grouping SM-Ama, SM-Sfr and SM-Hpo), SS and Sve (Fig. 8). Kappa and overall accuracy parameters varied among wetland sites (see Table 3). Producer's and user's accuracy parameters calculated for each plant community of interest across wetland sites showed high accuracy results for most plant communities, except for *Sve*, which was mainly confused with SS and SM communities (see Table 4). Plant communities abundances obtained by means of remote sensing were significantly correlated with those estimated by means of field sampling across wetlands ($R > 0.9$; $P < 0.01$; Fig. 9).

4. Discussion and conclusions

Our study comprises both basic and applied research by developing a rapid and effective technique to assess and monitor the ecological condition of semiarid Mediterranean wetlands. According to our results, the main objectives of the study were achieved: (1) plant community types were obtained in a systematic and reproducible way, (2) the proposed wetland condition index related wetland plant community composition and watershed hydrological pressures and (3) plant community maps obtained by means of remote sensing offered high accuracy and high spatial resolution in a wide range of wetland sizes. Our approach provides an effective procedure for using very high resolution airborne images of plant communities to evaluate the status of semiarid Mediterranean saline wetlands in relation to agricultural hydrological pressures at the catchment scale, thus serving as a tool for watershed management in order to prevent the actual wetland degradation rates (Zedler and Kercher, 2005; Dudgeon et al., 2006). The indicator

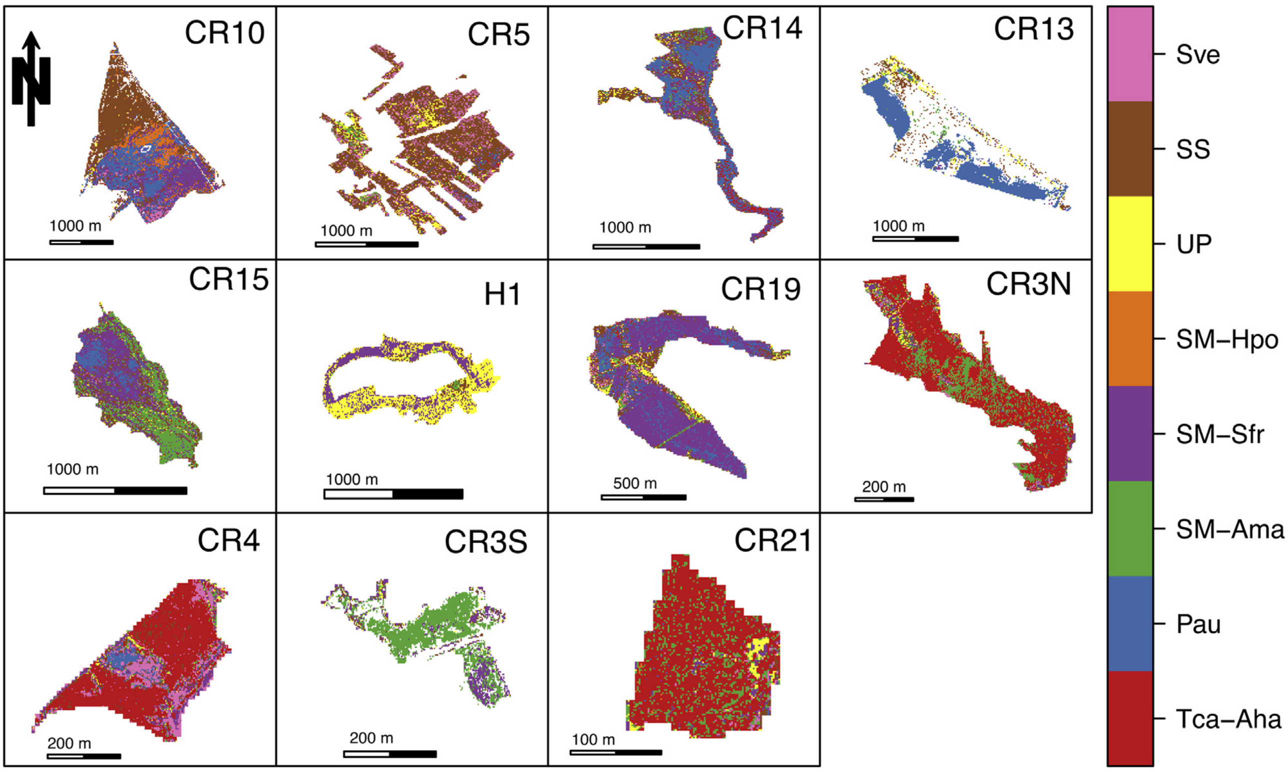

Fig. 8. Wetland plant community maps. Wetland keys: CR10: Carmoli; CR5: Alcanara; CR14: Ajauque; CR13: Lopoyo; CR15: Derramadores; H1: Rasall; CR19: Boquera de Tabala; CR3N: Cañada Brusca North; CR4: Matalentisco; CR3S: Cañada Brusca South; CR21: Sombrerico. Plant community keys: Sve: *Suaeda vera* dominated; SS: salt steppe; UP: dominated by upland plants; SM-Hpo: salt marsh dominated by *Halimione portulacoides*; SM-Sfr: salt marsh dominated by *Sarcocornia fruticosa*; SM-Ama: salt marsh dominated by *Arthrocnemum macrostachyum*; Pau: *Phragmites australis* dominated; Tca-Aha: *Tamarix* sp. and *Atriplex halimus* mixed community.

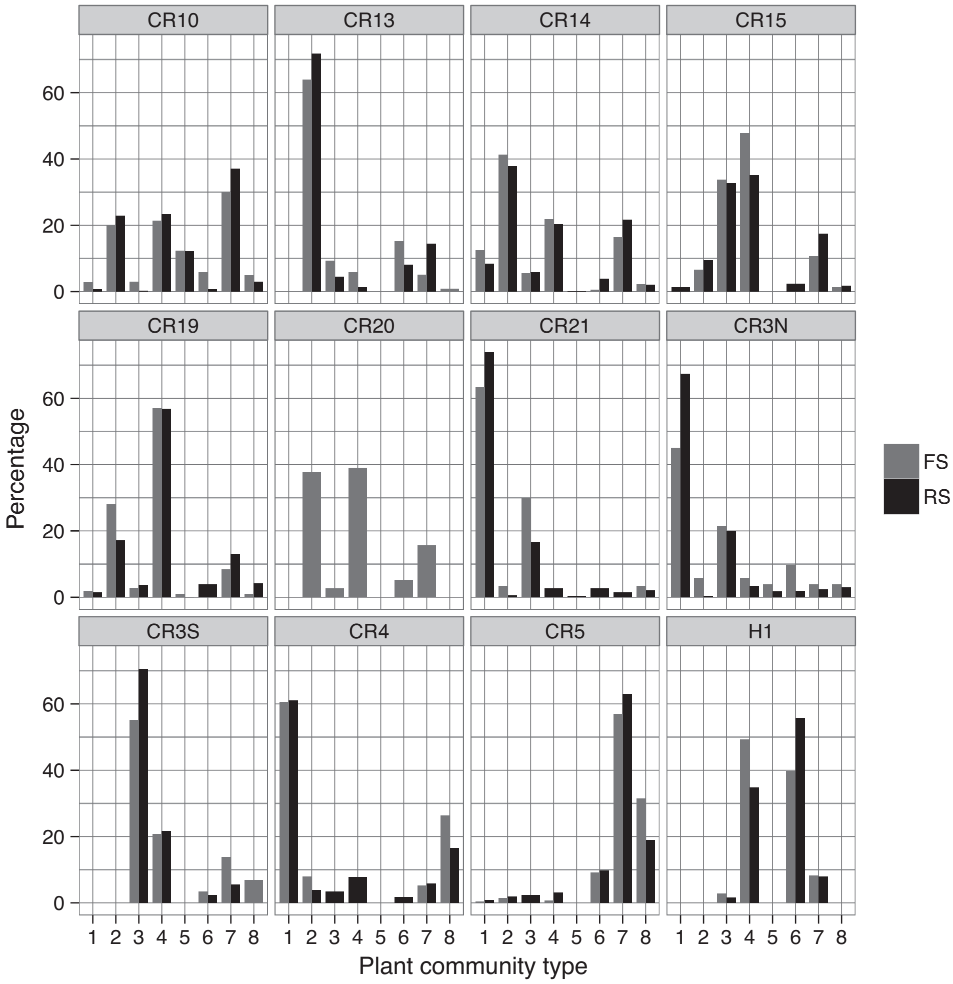

Fig. 9. Plant communities abundance in the study wetlands according to field sampling (FS) and remote sensing (RS) data. Wetland keys—CR10: Carmoli; CR13: Lopoyo; CR14: Ajauque; CR15: Derramadores; CR19: Boquera de Tabala; CR20: Playa de la Hita; CR21: Sombrerico; CR3N: Cañada Brusca North; CR3S: Cañada Brusca South; CR4: Matalentisco; CR5: Alcanara; H1: Rasall. Plant community keys—1: *Tamarix* sp. and *Atriplex halimus* mixed community (Tca-Aha); 2: *Phragmites australis* dominated community (Pau); 3: salt marsh dominated by *Arthrocnemum macrostachyum* (SM-Ama); 4: salt marsh dominated by *Sarcocornia fruticosa* (SM-Sfr); 5: salt marsh dominated by *Halimione portulacoides* (SM-Hpo); 6: dominated by upland plants (UP); 7: salt steppe (SS); 8: *Suaeda vera* dominated community (Sve).

wetland plant communities found in this study were coherent with indicator plant taxa obtained in previous studies for these type of wetland ecosystems and pressures (Martínez-López et al., 2012). Therefore, the study develops a cost-effective method for wetland regional assessments, thus overcoming the limitations of existing condition indices for semiarid Mediterranean saline wetland ecosystems (Ortega et al., 2004; Casta neda and Herrero, 2008).

Wetlands contain diverse habitats, which are prone to generate different plant species and communities due to intrinsic natural variability (Innis et al., 2000), while acting as refugees for some native wetland plants against invasion by alien plant species (Schwartz and Jenkins, 2000). Hydrological perturbations represent a major threat to biodiversity in wetlands, that contain plant communities adapted to a specific range of soil salinity and flooding duration. Salt marsh taxa and communities can endure sporadic floodings and are very salt-tolerant (Pujol Fructuoso, 2002; Vicente et al., 2007), while salt steppe species are less frequent at flooded areas (Caballero, 1999; Álvarez-Rogel et al., 2000). These ecosystems do no present substantial inter-annual variation in perennial taxa and plant community composition in absence of perturbations (Martínez-López et al., 2012). Furthermore, major differences in vegetation composition in semiarid Mediterranean wetlands are based on their relative abundances, rather than on taxonomic composition (Zedler et al., 1999), which is limited by abiotic conditions, such as soil salinity, anoxic conditions and waterlogging (Sanchez et al., 1998; Álvarez-Rogel et al., 2007a). However, the invasion of semiarid Mediterranean saline wetlands by *Tamarix canariensis*, *Atriplex halimus* and *Phragmites australis* seems to have altered characteristic plant communities, resulting in the replacement of valuable halophytes by more generalist and opportunistic taxa, probably due to decreased salinities and perturbed flooding regimes. *Phragmites australis* usually forms monospecific stands in areas with regular water flows, reducing species diversity, and therefore indicating wetland degradation (Burdick and Konisky, 2003; Álvarez-Rogel et al., 2007b; Tulbure et al., 2007), whereas *Atriplex halimus* in association with *Tamarisk canariensis* can form dense stands in areas with lower salinity conditions (MARM, 2009).

The fact that Matalentisco (CR4) and Boquera de Tabala (CR19) wetlands, both lacking of protection status, showed very contrasting ecological condition values due to watershed hydrological pressure reinforces the idea that watershed management plays an important role for wetland conservation (Fig. 7; Wigand et al., 1999; Turner et al., 2003; Mack, 2006). Another example is the contrasting plant community composition and condition index values of Cañada Brusca wetlands, whose North (CR3N) and South (CR3S) fragments clearly receive different surface hydrological influences (Fig. 7). Positive index values can be considered as indicative of a

Table 4
Error matrix calculated using all wetland plant community maps and validation sites. Plant community keys: Tca-Aha (*Tamarix* sp. + *Atriplex halimus*), Pau (*Phragmites australis*), SM (Salt marshes), SS (Salt steppe), Sve (*Suaeda vera*).

	Reference data						User's accuracy
	Tca-Aha	Pau	SM	SS	Sve	Total	
Map data							
Tca-Aha	**38**	5	11	4	3	61	62.30%
Pau	2	**106**	24	5	4	141	75.18%
SM	6	25	**175**	19	8	233	75.11%
SS	3	8	29	**135**	9	184	73.37%
Sve	4	7	9	10	**28**	58	48.28%
Total	53	151	248	173	52	**677**	
Producer's accuracy	71.70%	70.20%	70.56%	78.03%	53.85%		
Overall accuracy: 71%				Kappa: 0.34			

Bold values correspond to the number of correctly classified sites per plant community and to the total number of validation sites.

good/fair ecological status, where natural plant communities dominate the wetland site, while negative values indicate dominance of invasive plants and wetland degradation due to strong hydrological pressures.

According with this and previous studies (Belluco et al., 2006; Rooney et al., 2012; Mwita et al., 2013), remote sensors have been shown to be suitable for the study and monitoring of wetland plant communities. After our results, random forest models showed to be valid only within the spatial extent in which they were developed. High spatial resolution played an important role when mapping small wetlands, reducing also the within-pixel heterogeneity, and therefore increasing their spectral separability (Belluco et al., 2006). However, plant communities are not discrete entities and their species composition may vary slightly across wetlands. Therefore, intraclass heterogeneity, as well as inaccuracies related to georeferenced sampling plots in the field, may have affected classification accuracy, which was comparable to results obtained on similar recent studies (Chen and Lin, 2013). Plant community represented by *Suaeda vera* (Sve) showed the lowest accuracy values (Table 4) probably due to the fact that it is associated to perturbed areas like abandoned agricultural areas, which might increase heterogeneity among sampling plots.

Watersheds and wetland sites in the Southern part of Murcia province were among the most perturbed after the application of the watershed condition (WARP-IDW; Fig. 3) and the wetland condition (WCI_{PC}; Fig. 7) indices, respectively. Moreover, image classification of plant communities at these wetland sites showed generally less accuracy than in northern wetlands (Table 3). This might reflect the fact that their plant communities composition deviates from the characteristic ones due to the invasion of opportunistic taxa, which make them more heterogeneous and difficult to discriminate by means of remote sensing.

In this study, we only disposed of one single image and therefore we lacked of phenological information of vegetation, which would surely improve the discrimination of plant communities. Due to the increasing public availability of remotely sensed data, together with the use of free and open source software (Steiniger and Hay, 2009), the proposed wetland monitoring and condition assessment method can be easily extended and further developed in the Mediterranean region (Herrero and Casta neda, 2009; MacKay et al., 2009), preferably using several images representing different seasons (Xie et al., 2008) for image classification analysis. For this purpose, the use of other advanced remote sensing sources like present and forthcoming satellite sensors is advised, *e.g.* SPOT, Landsat-8 and Sentinel 2 missions (Drusch et al., 2012; Irons et al., 2012), since they offer multi-temporal images of medium and high spatial and spectral resolution, respectively.

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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.jag.2013.07.005.

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