Hydrobiologia (2012) 689:23–36
DOI 10.1007/s10750-011-0679-3

POND RESEARCH AND MANAGEMENT

Vegetation of Mediterranean temporary pools: a fading jewel?

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Received: 25 October 2010 / Accepted: 5 March 2011 / Published online: 21 April 2011

Abstract Although Mediterranean temporary pools are of great value for conservation, they are in rapid decline under the impact of various forms of anthropogenic pressure. Their disappearance from the landscape may result in a weakening of the biological connections between pools due to increasing isolation and the impoverishment of their communities. In Western Morocco (province of Benslimane), temporary pools have undergone severe regression over the past decades. The quantification of these losses and the impact on the richness of plant communities remain, however, unstudied. Since this is of vital importance for the conservation of the biodiversity of these habitats, a study has been undertaken associating (1) an assessment of the pool losses (both in density and surface area) between 1955 and 2001 using remote sensing, (2) surveys of vegetation and water depth (in 2006) in 48 pools, and (3) an assessment of the density and surface area of pools occurring within a 3 km radius around each of the sampled pools. The results show a loss of 23% in number and 61% in surface area of pools in the province over a period of 47 years. This decline, promoted by their small size and shallowness, is probably related to socio-economic changes (intensification of agricultural practices and population growth). The richness in characteristic and rare species of the pools was related to both local (water depth) and regional features (land use, pool density and total water surface area in the surrounding landscape). The significant impact of the current density of pools and their total surface area on the conservation value of the studied pools suggests a weakening of the metacommunity dynamics between pools. Given the rapid socio-economic changes in the province and the

Guest editors: D. Boix, B. Oertli, R. Céréghino, T. Kalettka, J. Biggs & A. P. Hull / Pond Research and Management in Europe – Proceedings of the 4th conference of the European Pond Conservation Network (Berlin 2010)

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current rate of pool disappearance (0.5% per year) we predict a continuing reduction in pool density with a high risk of the widespread loss of their unique flora in the long term.

Keywords Connectivity · Conservation value · Habitat loss · Remote sensing · Temporary wetlands · Vegetation

Introduction

Over the past decades, temporary wetlands have attracted the attention of researchers in theoretical and applied ecology (e.g. Braun-Blanquet, 1936; Wiggins et al., 1980; Deil, 2005; Williams, 2006). Their great diversity (ecological and biological) and their widespread distribution worldwide (Nicolet et al., 2004; Deil, 2005; Williams, 2006) have probably contributed to the privileged place they occupy today in aquatic research. Temporary wetlands play an important role in the landscape, such as flood control, renewal of groundwater, retention of toxic products and recycling of nutrients (Keddy, 2000; Williams, 2006). They also possess great value in terms of usage by society (e.g. water availability for man and wildlife, grazing, agriculture, harvest of medicinal plants, etc.) (Williams et al., 2004; Biggs et al., 2005) and constitute remarkable habitats for a unique flora and fauna contributing to the regional biodiversity (Williams et al., 2004; Biggs et al., 2005; Oertli et al., 2008).

Despite their value, temporary wetlands are rapidly declining worldwide (King, 1998; Nicolet et al 2004; Deil, 2005). Mediterranean temporary pools are among the most threatened ones because of their small size and shallowness (Grillas et al., 2004). During the past century, 94% of the pools disappeared in Spain (Azuaga Country) (Gallego-Fernandez et al., 1999). In the United States, 60 to 85% of pools were destroyed in California's Central Valley (Federal Register, 1994; King, 1998) and 97% in San Diego County (Weir & Bauder, 1990). In Northern European countries such as England, 75% of the pools that existed at the beginning of the twentieth century disappeared over the past 15 years and 1% of pools in Britain disappear each year under the impact of drainage, filling, agriculture and urban development (Williams et al., 1999). The loss of pools has often been associated with changes in the structure of the landscape, which in turn are linked to socio-economic factors and technological development (Gallego-Fernandez et al., 1999).

Although Mediterranean temporary pools have undergone perceptible regression, few studies have quantified this loss (Saber, 2006). Remote sensing constitutes an effective tool for identifying and monitoring the state of wetlands (Shuman and Ambrose 2003; Baker et al., 2006; Baker et al., 2007; De Roeck et al., 2008; Levin et al., 2009). However, its use remains restricted for temporary pools, because of their small size and their hydrological character (succession of dry and wet phases). To our knowledge, the few studies using remote sensing for temporary pools have focussed on the identification of potential breeding sites for amphibians (Doñana National Park in Spain: Gómez-Rodríguez et al., 2008), the determination of discrimination limits for pools in the landscape in function of annual rainfall and pool size (South Africa: De Roeck et al., 2008) or the monitoring of flooding regimes (Sandoz et al., 2004). Its application for the assessment of pool losses, at least in the Mediterranean region, appears to be poorly documented (Saber, 2006).

Pool communities function as meta-communities (Leibold et al., 2004; De Meester et al., 2005; Vanschoenwinkel et al., 2008a) and the biotic connections between pools are maintained by various dispersal mechanisms (both active and passive via vectors such as animals, water and wind) (Green & Figuerola, 2005; Vanschoenwinkel et al., 2008a, b). A decline in pool density in the landscape results in increased isolation and weakening of these biotic connections (Gibbs, 1993; Oertli et al., 2005). Therefore, the disappearance of pools is not only accompanied by a direct loss of local biodiversity (King, 1998; Oertli et al., 2005) but also by the impoverishment of biodiversity at a regional level (Amezaga et al., 2002; Zedler, 2003).

At the scale of the Mediterranean basin, Morocco stands out because of the high density of temporary pools through almost all of its territory, from the Atlantic plains to the eastern high plateaus and the mountainous regions (Thiéry, 1987). They are especially abundant in western Morocco, mainly in the province of Benslimane, where they exhibit a diversity of sizes, shapes, depths, usages and ecological characteristics (Thiéry, 1987; Rhazi et al., 2001). They harbour a great diversity of both flora (Rhazi et al., 2001; Grillas et al., 2004) and fauna (Thiéry, 1987). The temporary pools in this province are exposed to increasing anthropogenic pressures, such as drainage, agriculture, grazing and urban development. These pressures lead, depending on their intensity, to major (both biotic and abiotic) changes in the habitat or to its destruction (Grillas et al., 2004; Saber, 2006). Over the past two decades, the number of pools in the Benslimane province (as throughout the rest of Morocco) has undergone a perceptible, yet to be quantified regression. The conservation of these habitats and of their biodiversity in a context of global change requires knowledge of the patterns of change in their number and surface area and also of the impact of their usage and connectivity within the surrounding landscape on their specific richness at local and regional levels. In this context, the main aims of the present work are:

(1) To undertake an exhaustive inventory of the pools in the province of Benslimane using remote sensing and to analyse their spatio-temporal dynamics (1955–2001).
(2) To assess the relations between pool connectivity and land use and the species richness and conservation value of the pools.

Materials and methods

Study area

The present work was carried out in the province of Benslimane (255.000 ha) (Western Morocco), located on the Atlantic coast between Rabat and Casablanca (Fig. 1). The geological substratum is

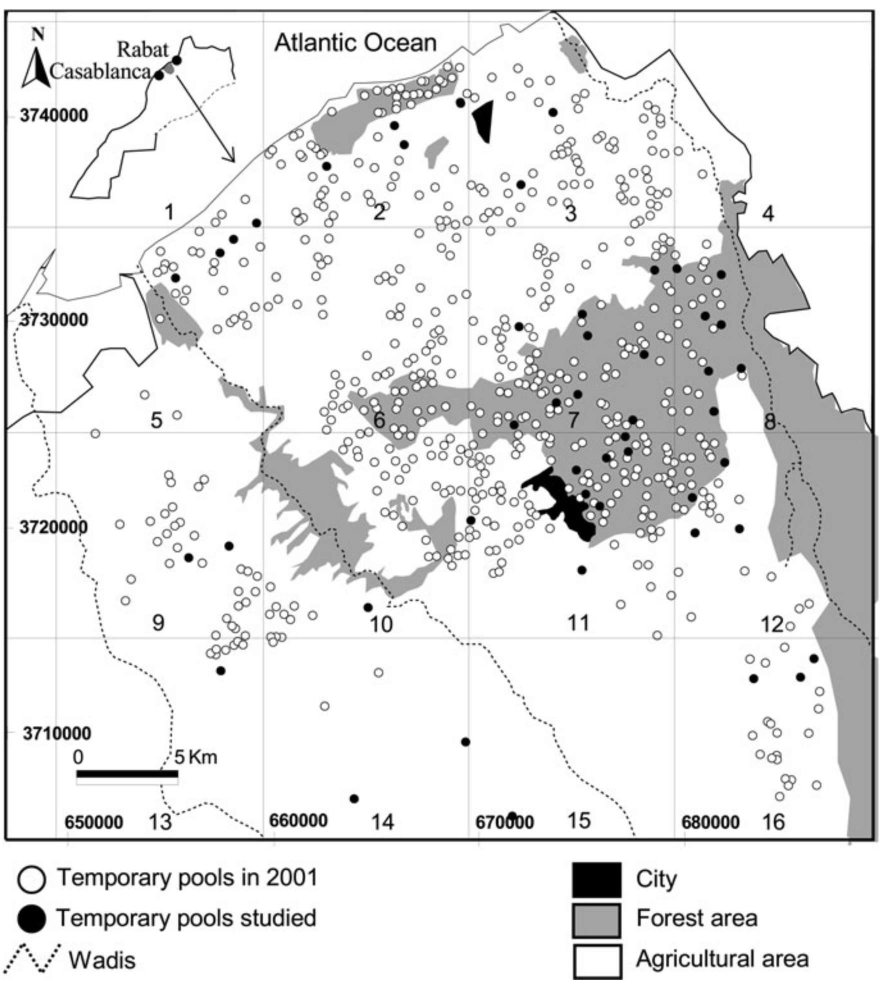

Fig. 1 Spatial distribution of temporary pools present in 2001 in the province of Benslimane. The numbers on the map correspond to quadrants of 100 km^2 used in the analysis of spatial losses of pools. The 48 study pools are indicated in *black*. For better visibility, the pools are represented by points. *Source*: Landsat satellite image of 20/01/2001

mainly constituted of quartzitic sandstone or schistose. The bioclimate, according to Emberger classification (1955), is Mediterranean sub-humid in the coastal area (about 7 km wide; average rainfall 465 mm per year; average minimum temperature is 7.1°C, and average maximum temperature is 27.3°C, Bouznika meteo station data 1960–2006) and semi-arid (rainfall: 450 mm per year; average minimum temperature: 7.5°C, average maximum temperature: 29.5°C, Benslimane meteo station data 1960–2006) in most of the province (Zidane, 1990). Rainfall occurs mostly in winter and the total precipitation shows large fluctuations between years (222–832 mm for the sub-humid area and 146–803 mm for the semi-arid area). The landscape consists of a mosaic of agricultural land, natural forest (*Quercus suber* and *Tetraclinis articulata*) and man-made forest (*Pinus* and *Euclalyptus*) and a large number of temporary pools representing 2% of the total surface area of the province (Rhazi et al., 2006). The pools are highly valuable for the local populations that use them for grazing and watering livestock, for washing clothes, for agriculture (edges) and collecting medicinal plants (Rhazi et al. 2001).

Spatio-temporal dynamics of temporary pool distributions

To follow up the spatio-temporal dynamics of the number and surface area of temporary pools in Benslimane province, a map of Théron & Vindt (1955) representing vegetation and temporary pools and two Landsat images (1987 and 2001) were used. These two images (spatial resolution of 30 m × 30 m) were recorded in January when the pools were flooded. The minimum surface area of pools detected at this resolution is 0.36 ha (4 juxtaposed pixels). The images were treated under *Idrisi* software and the number and surface area of pools were determined using *ArcView GIS*. The density of pools (number of pools/km^2) has been calculated for each of the three dates while differentiating for agricultural and forest environments. The number of extinct pools in the two periods: 1955–1987 and 1987–2001, as well as the number of pools present in 2001, have been inventoried in quadrants of 100 km^2 (16 in total) (Figs. 1, 2).

Vegetation study

In Benslimane, 48 temporary pools were selected according to land use in the surrounding area (forest: $n = 23$ or agricultural: $n = 25$) (Fig. 1). The selected pools were distributed homogeneously over the province to cover the different bioclimatic conditions (subhumid; semi-arid) reflecting continentality and geological (sandstone-quartzit; schist) gradients.

During two visits in 2006 (March/April and June 2006), which was a wet year with a total of 658 mm of rain (146% of the mean annual precipitation of Benslimane), the vegetation of each pool was studied. Since the species composition of the vegetation varies strongly within pools along hydrological gradients (Wilson & Keddy, 1985; Lenssen et al., 1999; Rhazi et al., 2001), two plots were surveyed per pool, one at the centre and one at the periphery. Phytosociological surveys were conducted on 81 m^2 relevés (9 × 9 m) homogeneous on the basis of their dominant species and potentially including several "associations" (sensu Braun-Blanquet). The surface area of the relevé is larger than the minimal area for the vegetation of Mediterranean temporary pools allowing for an almost complete list of plant species. On these relevés the coverage of each species was noted according to the Braun-Blanquet scale (Kent & Coker, 1992). Water depth (when present) was measured in each plot at each date. The pool size (circumference) of each of the 48 pools was measured in situ using a GPS.

The vegetation of each pool was characterized by the total species richness (cumulative number of species of both plots at both dates), the number of rare species of patrimonial value (according to Fennane & Ibn Tattou, 1998 for the vascular plants and Elkhiati, 1995 for macro-algae) as well as the number of species that are characteristic of pools and opportunistic species. The pool characteristic species were defined as aquatic plants and amphibious plants *sensu lato*, which are infeodated more or less strictly to temporary pools according to the studies of Chevassut & Quézel (1956), Nègre (1956), Médail et al., (1998) and studies on Moroccan temporary pools (Rhazi et al., 2006, 2009). The opportunistic species have been defined as terrestrial plants, generally frequent in the forest and agricultural surroundings (Maire, 1952–1987; Fennane et al., 1999; Fennane et al., 2007) which are peripheral to

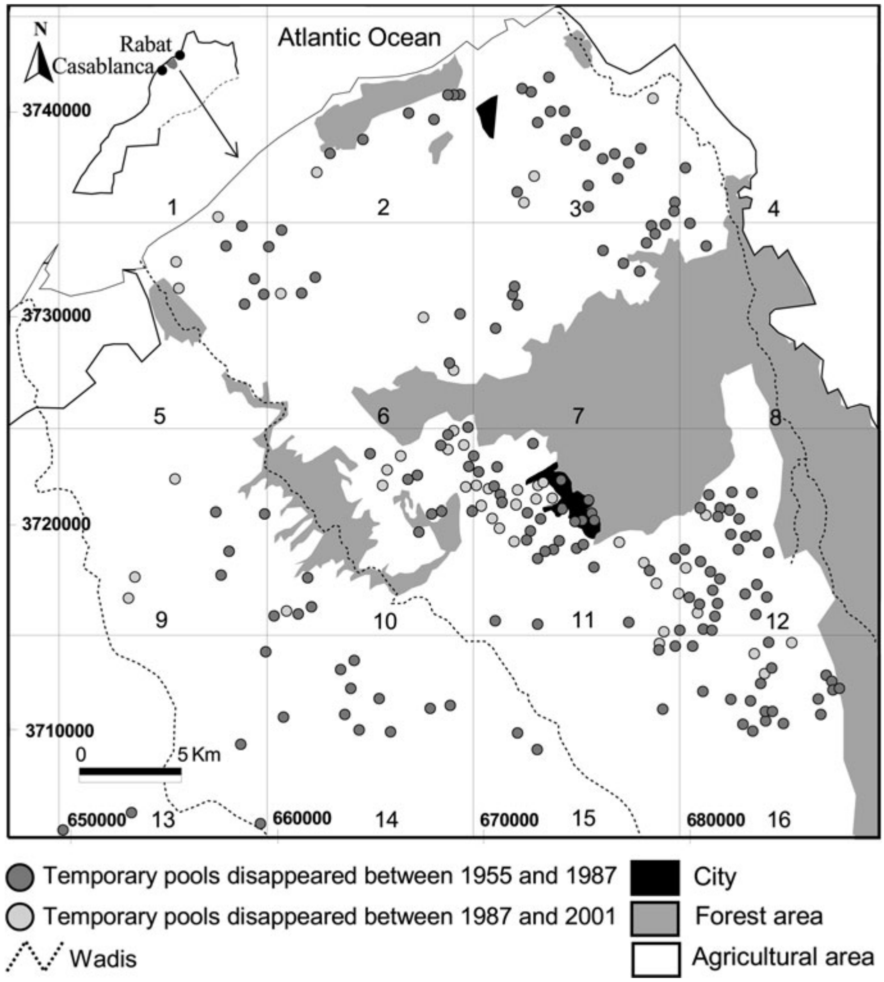

Fig. 2 Spatial distribution of the lost pools in the Benslimane province between 1955 and 1987 and between 1987 and 2001. The *numbers* on the map correspond to quadrants of 100 km^2 used in the analysis of the spatial losses of the pools. For better visibility, the pools are represented by points. *Source*: Vegetation map (1955) and Landsat satellite images (1987; 2001)

the pools, and penetrate the pools during the dry phase. The rare species of patrimonial interest constitute a subgroup of the pool characteristic species.

In order to study the effect of the degree of connectivity within the landscape on the species richness of the studied pools, the number and total surface area of the pools present within a 3 km radius around each of the 48 pools were calculated using *ArcView GIS*.

Data analysis

Spatio-temporal dynamic of the pools

Temporal variations (between 1955 and 2001) in the number and surface area of pools per quadrant of 100 km^2 in forest and agricultural environment have been tested using MANOVAs (repeated measures ANOVA). To analyse whether the temporal variation in pool surface area in agricultural environment was related to their size a MANOVA was used. The rate of disappearance of the pools per decade has been compared between the two periods (1955–1987 and 1987–2001) on quadrants of 100 km^2 by performing a MANOVA after a square root transformation of the data to become a normal distribution.

Pool characteristics and specific richness

Differences in maximum water depth, pool size, density and total surface area of pools within a radius of 3 km between pools in forest environment and those in agricultural environment have been tested with ANOVAs. The relation between maximum water depth and size of the pools was tested making use of a linear regression. Differences in individual abundances of species present in more than 4 pools out of 48 between the two pool types have been analyzed with Kruskal–Wallis tests. Differences in specific richness (total, pool characteristic species, opportunistic species and rare species) between the agricultural and forest pools were investigated using ANOVA. The effects of maximum water depth, pool size, pool density and total water surface within a 3 km radius on the species richness (total, opportunistic species, pool characteristic species and rare species) were studied using a multiple regression. All statistical analyses were performed in JMP4®.

Results

Spatio-temporal dynamics of pool distributions in the province of Benslimane

The inventory carried out at the three dates (1955, 1987 and 2001) showed a decline in both number and total surface area of temporary pools over time (Fig. 3). The total number of pools dropped from 871 in 1955 to 670 in 2001, a disappearance of 201 pools (23%) over a period of 47 years. During that same period, the total surface area of the pools dropped from 5086 ha in 1955 to 1994 ha in 2001, a loss of 3092 ha (61%) (Fig. 3). All lost pools were localised in agricultural areas (Fig. 4) and were concentrated for the most part around the city of Benslimane (Fig. 2). In contrast, no pools disappeared in forest environment (Fig. 4). The density of pools was similar for forest (3.8 per 10 km^2) and agricultural environments (4.4 per 10 km^2) in 1955, but had declined in agricultural environments to 2.9 per 10 km^2 in 2001, whereas it remained stable in forest environments (Fig. 4). The small increase in pool density in forest environments between the three dates (Fig. 4) is the result of a slight decrease in the surface area of forests without the loss of pools. The surface area of pools was significantly related to time ($F = 16.52$; df = 2; $P < 0.0001$), land use in the nearby environment ($F = 14.97$; df = 1; $P < 0.0001$) and their interaction ($F = 17.19$; df = 3; $P < 0.0001$). The decline in surface area over time was not significant for pools in forest environments ($P = 0.0898$), contrary to pools in agricultural environments ($P < 0.0001$). This decline in the surface area over time in agricultural environments depended significantly on pool size ($F = 103.25$; df = 2; $P < 0.0001$). The largest pools exhibited a greater regression (on average from 12.02 ha in 1955 to 2.57 ha in 2001) than the smaller pools (0.82 ha in 1955 to 0.76 ha in 2001). The rate of disappearance of pools per decade did not differ significantly between the two periods (1955–1987 and 1987–2001) ($P > 0.05$).

Plant species richness of temporary pools

For the whole set of 48 pools, a total of 253 species were found, comprising 185 annual (73%) and 68 perennial (27%) species. Eighty-five species were characteristic for pools (34%), among which fourteen rare species of patrimonial interest for Morocco were

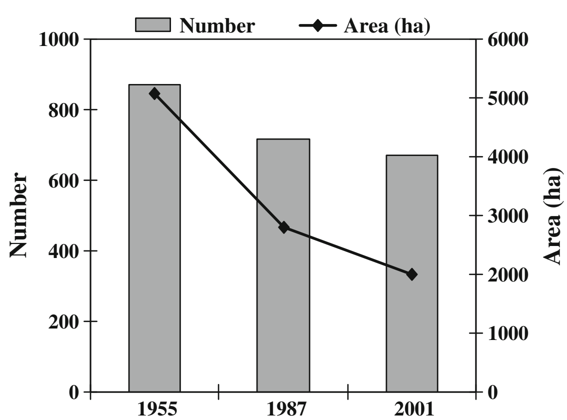

Fig. 3 Changes in the total number and surface area of temporary pools in the Benslimane province between 1955 and 2001

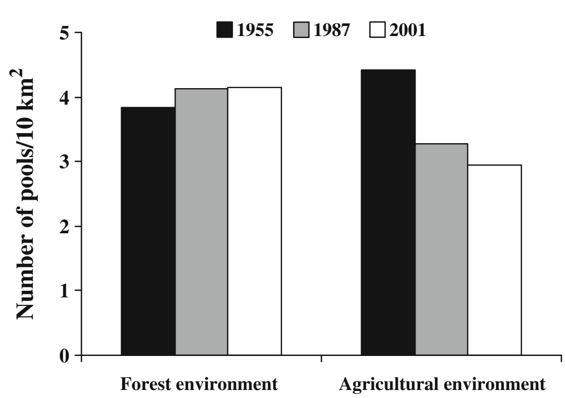

Fig. 4 Changes in the density of temporary pools (number of pools/10 km^2) between 1955 and 2001 in relation to the land use in their catchments

found (6%), including eight species that are endangered in North Africa according to the IUCN (Table 1). The remaining 168 species (66%) were opportunists and did not include any species of patrimonial value. The cumulative species richness for the forest pools was 205 and 210 for agricultural pools. The cumulative richness in pool characteristic species was similar for agricultural (72) and forest pools (73). In contrast, the cumulative richness in rare species was twice as high in forest pools (14) as in agricultural pools (6). The total richness in opportunistic species was similar for the two pool types (138 and 132, respectively, in agricultural and forest pools).

Of all the common species encountered in the survey, 53 were significantly more abundant ($P < 0.05$) in one of the two pool types (Table 2). The most abundant species in the forest pools (31) consisted mainly of pool characteristic species (23) (e.g. *Glyceria fluitans, Mentha pulegium, Eleocharis palustris, Isoetes velata, Pulicaria arabica, Juncus pygmaeus, Lythrum borysthenicum, Baldellia ranunculoides, Illecebrum verticillatum, Juncus tenageia*) while only eight species (e.g. *Bellis annua, Cistus salviifolius, Cistus monspeliensis, Tuberaria guttata*) were opportunists (Table 2). Conversely, the most abundant species in the agricultural area pools (22) were mostly opportunists (19) (*Plantago coronopus, Spergularia purpurea, Echium plantagineum, Diplotaxis catholica*, …) while only three were pool characteristic species (*Eryngium atlanticum, Lythrum tribracteatum, Verbena supina*) (Table 2).

Maximum water depth (range total 6–60 cm; Agricultural pools: 6–55 cm, Forest pools: 21–60 cm) and pool size (range total 0.4–7.6 ha; Agricultural pools: 0.4–7.6 ha, Forest pools: 0.6–5.5 ha) did not differ between the two pool types ($P > 0.05$). Both were weakly negatively intercorrelated ($r^2 = 0.12$; $P = 0.0169$). The density of pools within a 3 km radius was significantly higher for forest pools (34.48 ± 2.58) that for the agricultural pools (16.24 ± 2.16) ($F = 23.58$; df = 1; $P < 0.0001$), while the total water surface within that same radius did not show a significant difference between the two pool types ($P > 0.05$). Neither the total species richness per pool ($F = 2.69$; df = 1; $P = 0.1077$) nor the richness in opportunists ($F = 0.45$; df = 1; $P = 0.5063$) varied significantly between the forest pools (respectively, 61.60 ± 2.19 and 29.87 ± 2.45) and the agricultural pools (respectively, 56.56 ± 2.15 and 32.20 ± 2.46). In contrast, the richness in pool characteristic species ($F = 19.63$; df = 1; $P < 0.0001$) and in rare species ($F = 32.25$; df = 1; $P < 0.0001$) was significantly higher in forest pools (respectively, 31.74 ± 1.38 and 4.65 ±

Table 1 Check-list of rare or endangered species recorded in the studied pools with indication of their status of rarity (RR: very rare; R: rare; R? thought to be rare) as recorded in Fennane & Ibn Tattou (1998) and Elkhiati (1995) (for Characeae: *)

Species	Family	Current status	IUCN status for North Africa
Apium inundatum (L.) Reichenb. Fil.	Apiaceae	RR	VU
Elatine alsinastrum L.	Elatinaceae	RR	CR
Elatine brochonii Clavaud	Elatinaceae	R	VU
Exaculum pusillum (Lam.) Caruel	Gentianaceae	R	NT
Isoetes setacea Lam.	Isoetaceae	RR	CR
Isoetes velata A. Braun	Isoetaceae	R?	LC
Lippia nodiflora (L.) Michx.	Verbenaceae	R	LC
Lythrum baeticum Silvester	Lythraceae	R	VU
Lythrum thymifolia L.	Lythraceae	R	LC
Mariscus hamulosus (M. Bieb.) S.S. Hooper	Cyperaceae	RR	VU
Myriophyllum alterniflorum DC.	Haloragaceae	RR	LC
Oenanthe fistulosa L.	Apiaceae	R	LC
Pilularia minuta Durieu	Marsileaceae	RR	CR
Nitella opaca Ag.*	Characeae	R	–

The IUCN status (CR: Critically Endangered; VU: Vulnerable, NT: Near Threatened; LC: Least Concern) in North Africa is also given for each species (Rhazi & Grillas, 2010)

Table 2 Species showing a significantly different abundance between forest pools (F) and agricultural pools (A)

	Pool/opportunist	Frequency (%)	χ^2	*P*	Comparison
Glyceria fluitans (L.) R.Br.	P	85.4	15.13	*	F > A
Mentha pulegium L.	P	81.3	5.93	*	F > A
Eleocharis palustris (L.) Roem. & Schult.	P	79.2	4.23	*	F > A
Isoetes velata A. Braun	P	77.1	10.18	**	F > A
Pulicaria arabica (L.) Cass.	P	77.1	5.47	*	F > A
Lotus hispidus DC.	P	75.0	8.19	**	F > A
Juncus pygmaeus L.C.M.Richard.	P	68.8	9.56	**	F > A
Lythrum borysthenicum (Schrank) Litv.	P	66.7	4.51	*	F > A
Baldellia ranunculoides Lam.	P	62.5	10.91	**	F > A
Illecebrum verticillatum L.	P	60.4	5.69	*	F > A
Juncus tenageia L.f.	P	54.2	10.64	**	F > A
Isoetes histrix Durieu ex Bory	P	47.9	5.65	*	F > A
Exaculum pusillum (Lam.) Caruel	P	41.7	13.83	***	F > A
Ranunculus ophioglossifolius Vill.	P	41.7	8.02	**	F > A
Solenopsis laurentia (L.) C.Presl	P	37.5	7.43	**	F > A
Isolepis cernua (Vahl) Roem. & Schult.	P	35.4	7.39	**	F > A
Elatine brochonii Clavaud	P	27.1	18.43	***	F > A
Nitella opaca Ag.	P	27.1	9.96	**	F > A
Chara spp.	P	22.9	8.27	**	F > A
Apium inundatum (L.) Reichenb. Fil.	P	16.7	10.11	**	F > A
Myosotis sicula Guss.	P	14.6	8.67	**	F > A
Pilularia minuta Durieu	P	12.5	7.28	**	F > A
Isoetes setacea Lam.	P	10.4	5.93	*	F > A
Bellis annua L.	O	50.0	8.53	**	F > A
Briza minor L.	O	41.7	4.50	*	F > A
Asphodelus microcarpus Viv.	O	35.4	5.74	*	F > A
Cistus monspeliensis L.	O	33.3	24.54	***	F > A
Ornithopus compressus L.	O	25.0	4.75	*	F > A
Cistus salviifolius L.	O	25.0	12.44	***	F > A
Tuberaria guttata (L.) Fourr.	O	16.7	10.18	**	F > A
Lavandula stoechas L.	O	12.5	7.29	**	F > A
Eryngium atlanticum Batt. & Pit.	P	60.4	7.09	**	A > F
Lythrum tribracteatum Spreng.	P	33.3	13.51	***	A > F
Verbena supina L.	P	25.0	6.46	*	A > F
Plantago coronopus L.	O	77.1	7.63	**	A > F
Spergularia purpurea (Pers.) G.Don	O	64.6	6.27	*	A > F
Echium plantagineum L.	O	60.4	5.15	*	A > F
Diplotaxis catholica (L.) DC.	O	47.9	13.04	***	A > F
Cladanthus mixtus (L.) Chevall.	O	47.9	12.33	***	A > F
Rumex bucephalophorus L.	O	35.4	6.37	*	A > F
Medicago polymorpha L.	O	33.3	5.92	*	A > F
Hordeum murinum L.	O	31.3	5.65	*	A > F
Calendula arvensis (Vaill.) L.	O	27.1	15.85	***	A > F
Malva hispanica L.	O	27.1	4.45	*	A > F

Table 2 continued

	Pool/opportunist	Frequency (%)	χ^2	P	Comparison
Raphanus raphanistrum L.	O	27.1	16.43	**	A > F
Foeniculum vulgare Mill.	O	27.1	4.41	*	A > F
Scolymus hispanicus L.	O	25.0	10.38	**	A > F
Polygonum aviculare L.	O	22.9	5.15	*	A > F
Cichorium intybus L.	O	22.9	12.62	***	A > F
Arisarum vulgare Targ.Tozz.	O	18.8	9.87	**	A > F
Euphorbia helioscopia L.	O	16.7	4.77	*	A > F
Avena sativa L.	O	14.6	7.34	**	A > F
Kickxia spuria (L.) Dumort.	O	12.5	6.17	*	A > F

For each species the type Pool/Opportunist (P/O), the frequency over the whole set of pools (in %), the χ^2, the P value (* $P < 0.05$, ** $P < 0.01$, *** $P < 0.001$) and the comparison of abundance between the two types of pools are given

Table 3 Result of multiple regressions concerning the effect of the maximum water depth (D_{max}), pool size, pool density and total water surface within a 3 km radius on the species richness (total, opportunists, pool characteristic species and rare species)

Richness	Model	D_{max}		Area		Density of pools		Water surface	
	R^2	F	P	F	P	F	P	F	P
Total	0.10	3.07	0.0870	0.04	0.8379	1.48	0.2300	0.01	0.9356
Opportunist	0.29	**12.99**	**0.0008**	0.03	0.8637	0.08	0.7749	1.27	0.2660
Pool	0.44	**12.87**	**0.0008**	0.01	0.9406	**9.22**	0.0041	**5.74**	**0.0210**
Rare	0.34	**5.63**	**0.0222**	0.27	0.6055	**7.52**	0.0088	**4.33**	**0.0435**

The bold values highlight variables that show a significant effect ($P < 0.05$)

0.58) than in agricultural pools (respectively, 24.36 ± 0.97 and 1.28 ± 0.20). The total species richness showed no significant relation with the maximum water depth, size, density or total water surface within a 3 km radius (Table 3). The richness in opportunists was negatively correlated with the maximum water depth but not with the other variables (Table 3). The richness in pool characteristic species and rare species showed no significant relation with pool size, but both were positively correlated with the maximum water depth and with the density of pools and total water surface within a 3 km radius (Table 3). The effects of the interactions between these variables were non-significant ($P > 0.05$).

Discussion

Spatio-temporal dynamic of pools

The results of the present study evidenced a loss of 23% in number and 61% in surface area of the temporary pools in the province of Benslimane over a 47-year period (1955–2001). The discrepancy found between the loss in number and surface area probably reflects the greater human pressure on the large pools. The shallowness of these large pools (negative correlation between the maximum water depth and pool size) and their localisation in agricultural areas, where there have been major changes because of their status as private property, have probably facilitated their destruction. Indeed, the pool losses recorded in the province concern exclusively those in agricultural environments, whereas no loss was recorded in forest environments. At the present rate of destruction (0.5% per year), we expect that in less than 130 years, all agricultural pools in the province of Benslimane will have been destroyed. Moreover, we must acknowledge that these rates may be underestimated given the spatial resolution of the Landsat images (30 m × 30 m) which does not enable detection of small pools (minimum area of 0.36 ha). Nevertheless, such pools are relatively scarce in the province (Thiéry, 1987), which suggests that the margin of error is relatively low.

Although the loss of pools has affected the entire province (except the forest areas), the loss was mainly concentrated around the city of Benslimane and the south-eastern sector where there is a predominance of fertile black soil (Vertisols) that is very popular for agriculture (Ghanem, 1978). Urban development and agriculture are the main causes of destruction of wetlands (Gibbs, 2000) and temporary pools worldwide (Gallego-Fernandez et al., 1999; Williams et al., 1999; Nicolet et al., 2004; Zacharias et al., 2008). The disappearance of pools in Benslimane is probably related to socio-economic changes characterised by the intensification of agricultural practices (Machouri, 2005) and a high population growth rate. The population has increased by an average 1.9% per year between 1982 (138,437 inhabitants) and 2004 (197,704 inhabitants) (RGPH, 2004) to reach a density of 83 inhabitants/km^2. Poverty and the need for arable land of this growing population resulted in the destruction of pools (by drainage and/or filling) in order to create small farms. These socio-economic changes are similar throughout the southern Mediterranean countries (Plan Bleu, 2009) and are likely to increase in the future with the much more widespread destruction of temporary pools associated with the fragmentation and transformation of the landscape. The impact of such a loss of pools, both in number and surface area, on the species richness of plant communities of the temporary pools of Benslimane is currently unknown.

Plant species richness of temporary pools

The year in which we conducted the survey was a very wet year (146% of the mean annual rainfall for Benslimane) and therefore offered ideal conditions for the study of the vegetation. The species richness of the pools was studied cumulating the information gathered on 2 plots located, respectively, at the centre and the edge rather than on the total surface area of the pools. This choice was made to study the structure of the vegetation of the pools irrespective of their total size which could have introduced a bias in the analysis. Most of the species of each pool was found in the 2 relevés. The vegetation of the 48 sampled pools was very rich (253 species) with 34% of the species being characteristic for pools, 6% of which are rare in Morocco. Annual species were dominant in the vegetation (75%) which reflects their adaptation to fluctuating and relatively unpredictable environmental conditions (particularly the hydrology) imposed by this habitat (Williams, 2006). These constraints favour species with short cycles which invest more in sexual reproduction than in plant development (Zedler, 1987; Médail et al., 1998; Rhazi et al., 2006). The high species richness in these pools may be attributed to the alternation of flooded and dry phases characterising their hydrological cycle and the presence of persistent seed banks. Both allow for the coexistence and succession in time of numerous species (aquatic/amphibious and terrestrial) (Médail et al., 1998; Brock et al., 2003; Grillas et al., 2004; de Bélair 2005; Rhazi et al., 2006). The diversity in local characteristics (e.g. depth, water level, usages, etc.) of the pools also contributes to this richness (Fraga i Arguimbau, 2008). On both the northern and southern shore of the Mediterranean basin such high diversities have been recorded in temporary pool vegetation, for example in Sardinia (113 species in 9 ponds; Bagella et al., 2010), Minorca (360 species in 63 pools; Fraga i Arguimbau, 2008), Portugal (168 species in 29 pools; Pinto-Cruz et al., 2009), Morocco (155 species in 16 pools; Bouahim et al., 2010), Algeria (136 species in 26 pools; de Bélair 2005) and Tunisia (244 species in 36 temporary wetlands; Ferchichi-Ben Jamaa et al., 2010).

Effects of local habitat traits

A significant relationship between species richness and surface area was found in several studies on plant and animal communities of temporary pools (Spencer et al., 1999; Oertli et al., 2002; Eitam et al., 2004; Nicolet et al., 2004; Ruggiero et al., 2008). The high species richness in large pools is often attributed to their facility for colonisation and the availability of a diversity of microhabitats and thus of ecological niches which favour the coexistence of species with varied ecological requirements (e.g. opportunists and pool characteristic species) (Eitam et al., 2004). However, in our study, the total species richness and its components (pool characteristic, opportunistic and rare species) were not related with the size of the pools. This result could be explained by the fact that the surface area of the relevés was the same in all pools. However, in a previous study in the same research area and this time with complete census of the vegetation Rhazi et al. (2001), also found no significant relationship between plant species richness and the surface area of the pools. This lack of relationship could be explained by several factors: (i) in our study area the largest pools were shallow and widely used for grazing and agriculture, which may result in less favourable conditions for pool characteristic and rare plants which are sensitive to disturbances and demanding in terms of hydrology (Bouahim et al., 2010), (ii) the richness in pool characteristic and rare species was positively related to water depth which was higher in small pools; (iii) the small pools (0.4–1.4 ha) are large enough that size is not a limiting factor for plant richness (i.e. the range of size is small). In any case, the absence of a correlation highlights the need to conserve both large and small pools to maintain the aquatic biodiversity in the region.

Effect of land use

The use of the adjacent land had a significant impact on the richness in pool characteristic species and rare species but not on the total species richness or the richness in opportunists. Pools situated in forest environments harboured a greater richness in pool characteristic and rare species than those in agricultural environments. The much greater anthropogenic pressure in agricultural environments (culturing crops and grazing) compared to forest areas (only grazing) may explain this difference. Other studies have demonstrated that agriculture in neighbouring lands induces eutrophication of pools (e.g. use of fertilizers and herbicides) (Rhazi et al., 2001; Declerck et al., 2006; Angeler et al. 2008) and an increase in sedimentation rate that may cause burial of seed banks (Gleason et al., 2003; Devictor et al., 2007) and alter the hydrology (reduction of water depth). Pool characteristic and rare species are highly sensitive to physical (Sahib et al., 2009; Bouahim et al., 2010) and trophic disturbances (Rhazi et al., 2001) and our results showed that they were indeed most abundant in forest pools. The fact that a few species did show a greater abundance in agricultural pools (*Eryngium atlanticum, Lythrum tribracteatum, Verbena supina*) is probably linked to their specific life history traits (e.g. spines, production of numerous small seeds) making them more tolerant to disturbances linked with agriculture. These results highlight the interest of delimiting a buffer zone around pools in agricultural areas (Declerck et al., 2006) where certain activities (e.g. cultivation, use of fertilizers) would be banned to reduce negative impacts.

Effect of the connectivity in the landscape

The density and total water surface of pools in the landscape (3 km radius) did not affect the total species richness or the richness in opportunistic species. Opportunists (66% of all species) are terrestrial species that grow in neighbouring dry habitats (forests and agricultural lands). Their presence in pools is favoured by the dry phase (or a succession of dry years) and not by a dependence on aquatic habitats (Rhazi et al., 2009). Conversely, density and total water surface of pools in the landscape did have a positive effect on the richness in pool characteristic and rare species. An increase in pool isolation may limit dispersal and disturb the metacommunity dynamics. Previous and ongoing destruction of pools may already have lead or will lead to a loss of aquatic biodiversity, particularly of the remarkable species that are considered as 'jewels' of Mediterranean flora (Braun-Blanquet, 1936). Monitoring the density of pools and their aquatic vegetation (pool characteristic and rare species) is therefore indispensible to clearly identify the patterns of change and determine a threshold likely to disrupt healthy metacommunity dynamics.

Conclusions

The conservation value of the 48 studied pools is determined by local habitat traits (water depth), the land use (reflecting the human pressure) and the density and total water surface of pools in the adjacent landscape. The alteration of habitat traits through human activities (e.g. drainage, filling) or climatic changes and the degradation of the quality of the regional environment through continuous destruction of pools may result in the loss of their unique flora in the long term. Given the annual destruction rate of 0.5% and the increasing rate of socio-economic changes in the province, we predict a continuing decline in density and surface areas of pools in the landscape with a risk of limiting inter-pool connectivity. Long-term monitoring of the density of pools and their vegetation is indispensible in order to better determine the density-dependent effect on species that are of conservation priority. Remote sensing, as used in this research, appears to be an appropriate tool for pool density monitoring despite the error margin linked to the spatial resolution of Landsat images which does not enable detection of small pools. This could perhaps be corrected by comparing, over a few small areas, the number of pools inventoried on the ground with those detected using Landsat images.

The loss of pools, in number and surface area, highlights the gaps in knowledge of the functions and values of these habitats, both of the rural population and the decision makers (Dimitriou et al., 2006; Zacharias et al., 2008; Bouahim 2010). The high anthropogenic pressure on the pools and their decline in the Mediterranean region in general and in the Maghreb (Morocco, Algeria and Tunisia) in particular illustrates the urgency of introducing management and conservation measures for these habitats. Raising awareness in the general public and among decision makers of the value of these pools and the advantages that might be derived from their conservation is of primary importance for their long-term conservation.

Acknowledgments We thank Florence Daubigney for logistical and technical support, Nicole Yavercovski for helpful discussions, Pascale Nicolet for providing helpful documentation and the two anonymous reviewers for their remarks that improved the quality of this manuscript. This project has been achieved with the financial support of the Fondation Tour du Valat and Fondation MAVA.

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