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Long-term monitoring of an invasion process: the case of an isolated small wetland on a Mediterranean Island

Bruno Foggi[1]*, Lorenzo Lastrucci[1], Daniele Viciani[1], Giorgio Brunialti[2], Renato Benesperi[1]

[1] *Università degli Studi di Firenze, Dipartimento di Biologia Evoluzionistica, Biologia Vegetale, Via La Pira, 4, I-50121 Firenze, Italy; e-mail: bruno.foggi@unifi.it*

[2] *TerraData s.r.l. Environmetrics, Dipartimento di Scienze Ambientali, Università degli Studi di Siena, Via P.A. Mattioli, 4, I-53100 Siena, Italy*

Abstract: Invasions of *Typha* (cattail) and/or *Phragmites* (common reed) in wetland ecosystems result in changes in species richness, diversity and composition of vascular plants. These invasions are particularly harmful in lakes where threatened species and/or communities are found. The spread of two species of *Typha* (*T. angustifolia* and *T. latifolia*) and of *Phragmites australis* in the Stagnone Lake, on Capraia Island (Tuscan Archipelago – Mediterranean sea) was studied. We report this progressive invasion, documented by means of a series of vegetation maps (1991, 1995, 2000 and 2009). The expansion rate of the three invasive helophytes and the shrinking of the aquatic communities were studied using a GIS system. The impact of the spread of these three species on the floristic characteristics of the plant communities and the lake vegetation in general, was analysed by means of 15 plots of 1 m^2 in 2000 and in 2009. Statistical analysis of the two series shows a significant change in the floristic composition of the communities as a result of the invasion process. Many important groups of species, such as many aquatic species, decrease in number and in cover value.

Key words: *Typha*; *Phragmites*; conservation; GIS; multivariate analysis; plant diversity

Introduction

According to Elton (1958) "biological invasions are so frequent nowadays in every continent and island, and even in the ocean, that we need to understand what is causing them and try to arrive at some general viewpoint about the whole business". During the last 50 years the Mediterranean landscape has been subject to several variations in its vegetation cover due to changes in socio-economic circumstances, a phenomenon that has affected most of the Mediterranean islands (Delanoe et al. 1996). These changes have principally resulted in a transition from an ager-saltus-sylva based land use (Blondel & Aronson 1999) to an economy based on tourism, involving numerous short term visitors to the islands (Delanoe et al. 1996). This change makes the island ecosystems more prone to the invasion of alien species. These invasions by native or non-native species lead to a homogenization of regional flora that could be considered a principal cause of local extinction (Wilkinson 2004). According to Ricciardi & Rasmussen (1999), introduced species are the second most important cause of species extinction after habitat destruction. Dramatic impacts can occur when an introduced species functions as a keystone or engineer species and causes changes in the chemical and physical properties of the environment, in the flora and fauna, in the food chain, or in the structure of the invaded habitat (Crooks 2002). Invasions of *Typha* spp. generate dense stands with low plant diversity (Green & Galatowitsch 2001; Ehrenfeld 2003; Bowles & Jones 2006; Boers et al. 2006) and present a significant threat to wetland biodiversity. Invasion by *Typha* could lead to the loss of native species and alter ecosystem functioning, particularly affecting nutrient cycling, hydrology, and fire regimes (Vitousek & Walker 1989; Brooks et al. 2004; Callaway & Maron 2006; Boers et al. 2006). Similarly, looking at invasion by *Phragmites australis*, Lenssen et al. (2000) found a significant depression of floristic diversity in vegetation stands dominated by this species. Invasion of wetlands generally causes a significant change in the ecological conditions for aquatic species, giving rise to a reduction in their diversity. The spread of invasive species, both native and non-native, is common in several regions and habitats in Italy but, with the exception of some technical reports, has not so far been studied in detail, especially regarding the impact on local plant diversity of small wetlands. The problem of invasion by native or non-native species is particularly dramatic in island ecosystems (Simberloff 1995; Whittaker & Fernández-Palacios 2007) where human mediated-dispersal breaks the historical natural isolation of the island. The Stagnone Lake is the only permanent wetland in the Tuscan Archipelago and its ecosys-

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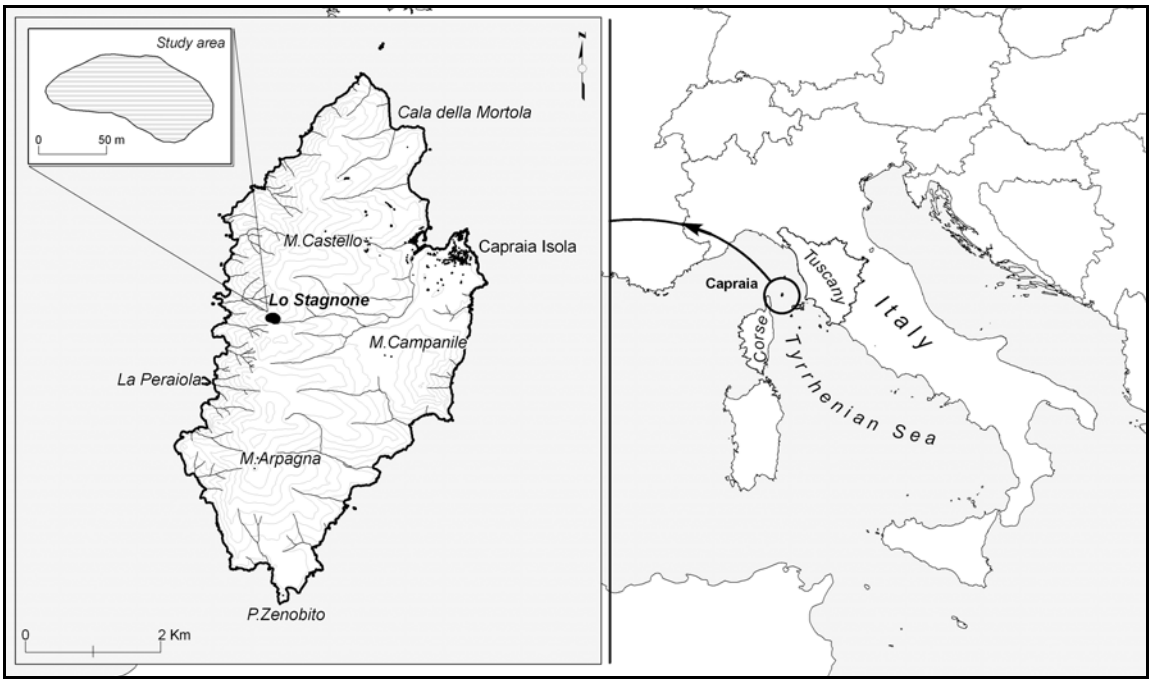

Fig. 1. Study area. Mediterranean Sea with the position of Capraia Island. In the detail Capraia Island with Stagnone Lake is shown.

tem contains several plant species such as *Ranunculus peltatus* subsp. *baudotti*, *Myriophyllum alterniflorum*, *Baldellia ranunculoides* and aquatic communities that are rare and threatened in the north Mediterranean region (Foggi & Grigioni 1999; Foggi et al. 2001). For this reason, this lake has been declared a SAC (Special Areas of Conservation) according to the "Habitat Directive" (CEE 83/43). The aim of the present study was to understand the changes occurring in a small lake following the invasion of highly competitive species such as cattails and common reed.

The overall objectives of our research were to document the invasion and spread of cattails and common reed, the resulting impact on the structure and plant diversity of the lake, and the effect of the invasion on the species of conservation interest.

Material and methods

Study area

Capraia Island is in the Tuscan Archipelago (north central Mediterranean), 53 km from the Italian coast and 26 km from the island of Corsica (Fig. 1). It has an area of 19.72 km^2; the maximum length is 8 km and the maximum width is about 4 km (Foggi & Grigioni 1999). The landscape is mostly mountainous, except for two small flat areas located in "Piana dello Zenobito" and "I Piani". The highest mountain is Monte Castello (445 m), which is a part of the N-S oriented mountain range (Foggi & Grigioni 1999). Capraia is made entirely of volcanic rocks originating from two different eruptions; several alluvial areas are present near the Porto Vecchio bay and in the flat area of "I Piani". The climate is typically Mediterranean with hot, dry summers and wet winters. According to Foggi & Grigioni (1999) the vegetation of Capraia comprise several types of maquis dominated by *Erica arborea* and garrigues dominated by *Cistus monspeliensis*; *Quercus ilex*-woodlands are nowadays absent on the island. The Stagnone Lake is situated at an altitude of 330 m in a small saddle. The deepest part of the lake is 140 cm and the water in it derives exclusively from precipitation. The study area covered a surface of 5,023.22 m^2 (Fig. 1 detail). The lake is under invasion by three species: *Typha angustifolia*, *T. latifolia* and *Phragmites australis*. The presence of *T. angustifolia* and *P. australis* on Capraia island was first recorded in 1839 by Moris & De Notaris (1839) near the Porto and confirmed by our personal observations from 1995, with the addition of *T. latifolia*. Of the three, only *T. latifolia* could be considered as non-native, being never previously reported for Capraia Island. The invasion of Stagnone Lake probably started some years after 1991. At this time the lake was covered only by aquatic communities dominated by *Ranunculus peltatus* subsp. *baudotti*. *Typha angustifolia*, *T. latifolia* and *Phragmites australis* were not recorded on the lake at this time. By 1995, more than one third of the lake surface was covered by a mixed community of *T. angustifolia* and *T. latifolia*. More recently, probably during 2006 or 2007 (because in 2005 we did not observe *P. australis* and in 2009 we see plants two-three years old), this first invasion of *Typha angustifolia* and *T. latifolia* was followed by a second one dominated by *P. australis*. All three species are able to spread rapidly by vegetative propagation (Haslam 1971; Galatowitsch et al. 1999) and this was probably the system used by these species during the invasion of the Stagnone Lake.

Vegetation maps

Variation in the extent of the plant-communities of Stagnone Lake was studied by means of a ArcGis software (ESRI'ArcGis, 9.2). Georeferenced aerial orthophotos (1995, 1998 and 2009) and several un-georeferenced photos dated 1991 were examined. All the orthophotos and the photo were re-examined and re-digitized to ensure consistency between them. All the vegetation types derived from the orthophotos (1995, 1998 and 2009) as seen in the computer-video were checked and mapped in the field using GPS. The vegetation types here used were "physiognomic vegetation types", for

the phytosociological interpretation of the plant communities of Stagnone Lake see Lastrucci et al. (2010). Boundaries between the vegetation types so recognized were digitalized using GIS. Four vegetation maps were derived and drawn at a scale of 1:500. The areas of the vegetation types identified were calculated for each of the 4 years (1991, 1995, 1998 and 2009).

To measure the spatial heterogeneity of vegetation cover the Simpson index was used (Reed et al. 2009).

Sampling design for floristic study

The study area was rasterized with a 5 ×5 m grid, and during the year 2000, 15 squares were selected at random. In the center of each selected square, a plot of 1 × 1 m was established. For each 1 × 1 m plot, the entire flora was recorded. Cover values were estimated in the field using the Braun-Blanquet scale (1932; see also Tulbure et al. 2007). According to Wild et al. (2004) and Bímová et al. (2004) the original Braun-Blanquet scale values were transformed as follows: 5 = 87.5%; 4 = 62.5%; 3 = 37.5%; 2 = 15%; 1 = 2.5%; + = 1%; r = 0.1%.

In the tables species with conservation value according to Regional Law 56/2000 of the Tuscan Region Administration are highlighted with the symbol *.

The field samples were carried out in May 2000 and 2009.

Data analysis

An Indicator Species Analysis (ISA: Dufrêne & Legendre 1997) was adopted to determine how strongly each species was associated with the year of sampling (2000 survey: 15 plots × 19 species; 2009 survey: 15 plots × 25 species). For each species, the Indicator Value (INDVAL) ranges from 0 (no association) to 100 (maximum association). The statistical significance of INDVAL was tested by means of a Monte Carlo test (10,000 iterations). Species data were square root transformed before the analyses, to downweight the species with higher cover values.

Compositional differences between the two surveys were tested using multi-response permutation procedures (MRPP). The Sřrensen-index was used as a distance measure and for rank transformation of the distance matrices. The separation between groups was calculated as the chance-corrected within-group agreement (A) and the p value was used for evaluating how likely an observed difference was due to chance ($A = 1$ indicates perfectly homogenous groups, while $A = 0$ indicates within-group heterogeneity equal to that expected by chance). In community ecology, values for A are commonly below 0.1, even when the observed data differs significantly from the expected (McCune & Grace 2002).

The changes in pattern of species composition were also visually evaluated using non-metric multidimensional scaling (NMDS; "slow and thorough" autopilot mode, quantitative version of Sørensen distance measure; McCune & Grace, 2002). The procedure was performed with 40 runs of real data, compared to 50 runs of randomized data, with 400 iterations each. Stress levels differed significantly ($p < 0.05$). A final 3-dimensional solution was selected (stress 11%, instability 0.00009).

To compare the two situations (2000 vs 2009) we also used four indices: species richness as the number of species for each plot, Shannon's index, Simpson's index and Pielou's index of equitability. The Ellenberg indicator values: light, moisture and nitrogen (Ellenberg 1979; Pignatti 2005) were used to test if the differences in floristic composition reflected variation in the various species ecological requirements. A Wilcoxon test was used to test if the differences between the two data sets were significant.

For statistical analyses we used PC-Ord (McCune & Mefford 1999) and STATISTICA (Statsoft 2006).

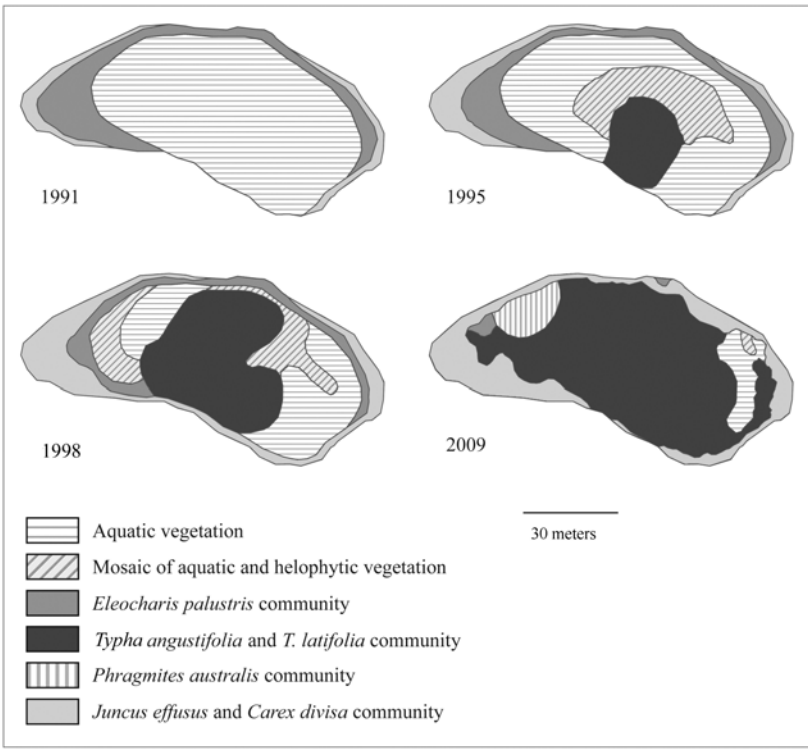

Fig. 2. Vegetation maps of the years studied.

Table 1. Percentages of the lake surface covered by the different physognomic vegetation types in the four survey years. Simpson Indices are calculated for the vegetation type cover values for the four years.

	1991	**1995**	**1998**	**2009**
Aquatic vegetation	73.44	45.51	24.36	5.78
Mosaic of aquatic and helophytic vegetation	–	15.92	13.48	0,50
Eleocharis palustris community	18.89	15.45	10.18	1.03
Phragmites australis community	–	–	–	5.40
Typha angustifolia and *T. latifolia* community	–	12.01	33.33	63.94
Juncus effusus and *Carex divisa* community	7.68	11.11	18.65	23.35
Simpson Index	0.3951	0.7053	0.7566	0.5008

Results and discussion

Vegetation dynamics

Vegetation maps (Fig. 2) of the study years show the changes occurring in the area and distributional pattern of the plant communities from 1991 to 2009. In Table 1 the percentages of lake surface covered by the different vegetation types in the 4 years are shown with the respective Simpson indices. The surface of *Typha* spp. communities expanded 214 m^2/year during the first period of invasion (from 1995 to 2000) and 170.8 m^2/year in the second part from 2000 to 2009.

Our results show a clear change in the vegetation cover of the lake: the *Typha*-community has increased from 0 to 63.4% coverage in 18 years; and common reed probably started its invasion during 2007 and by 2009 already covered 5.4% of the lake surface. In contrast, the aquatic communities dominated by *R. peltatus* subsp. *baudotii,* alone or with *Myriophyllum alterniflorum,* lost the greater part of their coverage. We predict that this process will continue until the lake is fully covered by a *Typha-Phragmites* community and all the species previously found there are lost.

Floristic variation

Table 2 shows the quantitative and qualitative floristic data from the 2000 and 2009 surveys. A total of 29 species were found. 19 and 25 species were found in 2000 and 2009, of which 14 were common to both surveys.

The Indicator Value (IV > 25) shows that three species were found specific to the 2000 survey (*Ranunculus peltatus* subsp. *baudotii, Alisma plantago-aquatica* and *Eleocharis palustris*); while 5 species were specific to the 2009 survey (*Lemna minor*, *Typha angustifolia*, *T. latifolia*, *Cyperus longus* subsp. *badius*, *Juncus effusus*). Some aquatic species present in 2000 (*Callitriche stagnalis* and *Potamogeton crispus*) were not found in the 2009 relevés. On the other hand several new species typical of terrestrial habitat were recorded: *Trifolium nigrescens*, *Sonchus asper*, *Rumex acetosella* and *Cynodon dactylon.*

The process of substitution of aquatic species from the terrestrial is also shows by the ISA results and by the MRPP that revealed a significant difference between the specific community composition in the two sampling years ($A = 0.05$; $p < 0.01$). Despite this fact, the small A statistic indicates that there is a broad overlap between the two communities.

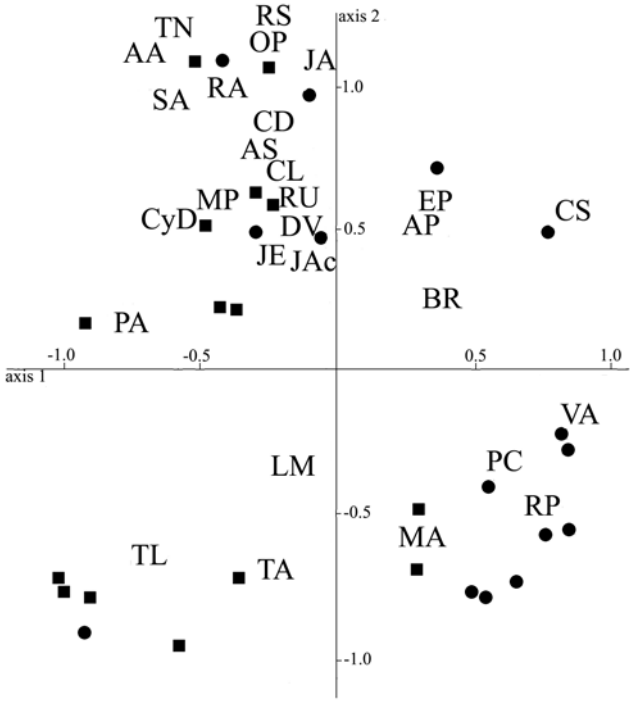

Fig. 3. Distribution of samples and species in the biplot of the NMDS analysis. ● – samples in 2000; ■ – samples in 2009. For the abbreviations of species see Table 2.

The first three axes of the NMDS ordination explained 87.5% of the total variation in species composition (41.7% for the first axis 1; 27.7% for the second axis and 18.1% for the third axis). In Fig. 3 we show the first two axes of the variation in samples and species. Plots sampled in the two surveys were clearly separated along axis 1 (73% of 2000 plots for positive values; 87% of 2009 plots for negative ones) in relation to a clear gradient in plant communities, from one typical of an aquatic habitat (positive half) to one more terrestrial in nature (the negative half). The quadrants lump the species according to the communities shown in the vegetation maps. In the first quadrant we found species of the community dominated by *E. palustris* that grows in the shallow water at the edge of the lake. In the second quadrant we found the true aquatic species. In the third quadrant the species comprising the first invasion become apparent as well as the *L. minor* which appeared after 2000. Finally, in the fourth quadrant we see the new arrivals: common

Table 2. Comprehensive list of the species surveyed, with their abbreviations and species occurrence percentages. Data are the percentage presence in the 2000 and 2009 surveys. Medium cover values in 2000 and in 2009 are given as IV = INDVAL.

Species	Abbreviation	Year	Species occurrence (%)		Indicator Value (IV)	IV from randomized groups (mean± SD)	*P*
			2000	2009			
Ranunculus peltatus Schrank subsp. *baudotii* (Godr.) C.D.K.Cook *	RP	2000	60	20	48.09	28.7±7.22	**0.0130**
Alisma plantago-aquatica L.	AP	2000	46.7	40	36.8	31.2±7.78	0.2150
Eleocharis palustris (L.) Roem. & Schultes *	EP	2000	40	6.7	27.1	19.9±6.76	0.2220
Myriophyllum alterniflorum DC. *	MA	2000	33.3	13	16.3	20.5±7.4	0.7360
Callitriche stagnalis Scop.	CS	2000	6.7	–	6.7	6.7±0.2	1.0000
Juncus articulatus L.	JAr	2000	6.7	–	6.7	6.7±0.2	1.0000
Potamogeton crispus L.	PC	2000	6.7	–	6.7	6.7±0.2	1.0000
Veronica anagallis-aquatica L.	VA	2000	6.7	–	6.7	6.7±0.2	1.0000
Baldellia ranunculoides (L.) Parl. *	BR	2000	6.7	6.7	3.3	8.4±5.0	1.0000
Oenanthe pimpinelloides L.	OP	2000	6.7	6.7	3.3	8.0±5.0	1.0000
Ranunculus sardous Crantz	RS	2000	6.7	6.7	5.1	8.9±4.1	1.0000
Lemna minor L.	LM	2009	–	73.3	73.3	26.8±7.1	**0.0010**
Typha angustifolia L.	TA	2009	26.7	53.3	33.3	30.7±8.3	0.3360
Cyperus longus L. subsp. *badius* (Desf.) Murb.	CL	2009	–	26.7	26.7	13.0±6.1	0.1090
Typha latifolia L.	TA	2009	33.3	40	27.5.	27.8±7.7	0.4200
Juncus effusus L.	JE	2009	46.7	46.7	27.2	32.9±7.8	0.7350
Agrostis stolonifera L.	AS	2009	13.3	20	16.5	15.8±5.9	0.4000
Carex divisa Huds.	CD	2009	13.3	26.7	14.3	17.4±6.5	0.6450
Rumex conglomeratus Murray	RC	2009	26.7	26.7	15	21.2±6.9	0.8980
Dittrichia viscosa (L.) Greuter	DV	2009	6.7	20	13.1	13.2±5.7	0.6070
Rubus ulmifolius Schott	RU	2009	13	20	13.1	15.7±6.0	0.6290
Anagallis arvensis L.	AA	2009	–	6.7	6.07	6.7±0.2	1.0000
Cynodon dactylon (L.) Pers.	CyD	2009	–	6.7	6.7	6.7±0.2	1.0000
Juncus acutus L.	JA	2009	–	6.7	6.7	6.7±0.2	1.0000
Mentha pulegium L.	MP	2009	–	6.7	6.7	6.7±0.2	1.0000
Phragmites australis (Cav.) Trin. ex Steudel	PA	2009	–	6.7	6.7	6.7±0.2	1.0000
Rumex acetosella L.	RA	2009	–	6.7	6.7	6.7±0.2	1.0000
Sonchus asper (L.) Hill	SA	2009	–	6.7	6.7	6.7±0.2	1.0000
Trifolium nigrescens Viv.	TN	2009	–	6.7	6.7	6.7±0.2	1.0000

reed and some species typical of more terrestrial habitats.

Data of species richness and that obtained according to the three indices used (Shannon, Simpson and Equitability) was not significant ($p > 0.05$).

The Ellenberg indicator values show a significant increase of the species not related to aquatic habitats (Wilcoxon test for the moisture indicator value $p < 0.01$).

According to Whyte et al. (2008), the arrival and expansion of *P. australis* seems correlated to a drift of lake ecosystems into a more terrestrial condition with a lesser influence of the water body and reduced numbers of species and individuals of species related to an aquatic habitat.

In the same way the presence of *L. minor* in the *Typha latifolia*-*T. angustifolia* and/or *P. australis* stands may be explained by the increasingly sheltered environment created by the presence of this emergent vegetation (Whyte et al. 2008) although, in our case, the Ellenberg indicator values for light was not significantly altered in our study ($p > 0.05$).

Conclusion

T. angustifolia and *P. australis* were first recorded on Capraia island in 1839, in the Vado del Porto, near the port. It appears unlikely that the invasion of the Stagnone Lake resulted from these individuals because they were present on the island for centuries before the first records for the Stagnone. Once started, the invasion of the lake proceeded very rapidly, especially in the initial phases, aided by the facilitation effect that the old individuals of cattail exert on the young plants (see also Callaway & King 1996). Invasion of these great helophytes has been reported several times in freshwater wetlands in North America (Wilcox et al. 1985; Galatowitsh et al. 1999; Rice et al. 2000; Choi & Bury 2003; Chun & Choi 2009) where in addition to *T. latifolia* and *T. angustifolia,* the hybrid between the two, *T.* × *glauca*, was reported. These species were reported as "weedy" invasive species (Haslam 1971; Gleason & Cronquist 1991). After a first phase dominated by *T. latifolia* and *T. angustifolia* follows the arrival of *P. australis* that tends to becomes dominant. How the shift of the lake to more terrestrial conditions had work is unknown but we can assume that the first arrival of the two cattails could have determine higher rates of evaporation faciliting its spread and the stabilization of *P. australis* as already observed by Wilcox et al. (1985) and Choi & Bury (2003). This process lead to the "terrestrialization" of the lake, process signed by the presences of several species not related to aquatic condictions.

Many managers report a loss of avian diversity in areas invaded by several species of *Typha* (Apfelbaum 1985, Galatowisch et al. 1999) and/or *P. australis* (Benoit & Askins 1999). There are very few quantitative studies on this issue, especially in the Mediterranean Region, and further studies seem desirable.

From a floristic point of view, several studies report a loss of diversity during the invasion of a wetland ecosystem (Green & Galatowitsch 2001; Ehrenfeld 2003; Bowles & Jones 2006; Boers et al. 2006). Our data do not show such a loss in diversity but this is due to the loss of species with conservation value (according to Habitat Directive – ECE, 93/43 and the Law on Biodiversity of the Tuscany Administration – LR 56/2000, Regione Toscana) and the arrivals of common species with no conservation value. The loss of rare plant species and communities is supported by our study, but additional work will be necessary to fully understand the ecological role of the lake in the wider Capraia ecosystem.

Figure 3 shows the phases of the invasion: it is possible that if the invasion continues, the common reed community will dominate the entire lake and the aquatic communities will be completely lost. The administration of the "Tuscan Archipelago" National Park has decided to completely restore the lake, and this study was preparatory to that decision. However, the problems of invasion will remain if a new culture regarding the use of alien species is not adopted by the inhabitants of these islands. Many examples of invasion and eradication were found in the literature (i.e. Keller 2000; Bernardoni & Casale, 2000) and strategies that prevent invasion rather than cure the ill effects, seem the better choice. As Simberloff (2003) says "I believe the strongest ethical bases, and possibly the only one ethical base, for concern over introduced species are that they can threaten the existence of the native species and communities and they can cause staggering damage, reflected in economic terms, to the human endeavour". Accordingly, particular attention should be focussed on the introduction of species that could have invasive characteristics, especially when introduced to islands. In these cases the philosophy of "guilty until proven innocent" should be the first principle to follow.

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