

# *Acacia cyclops* A. Cunn. ex G. Don (Leguminosae) in Italy: first cases of naturalization

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## Abstract

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The first two cases of naturalization of *Acacia cyclops* are reported for Italy. Young trees were observed growing in the wild some 15 years ago on Linosa (Pelagie Islands, Strait of Sicily). A decade later, this alien plant should no longer be considered as a casual, since a very intensive process of self-sown regeneration has been observed in some plantations on Lampedusa, the major island of the same Archipelago. The available literature suggests the need for careful monitoring of the ongoing invasion process, as *A. cyclops* has already shown a very invasive behaviour elsewhere within Mediterranean-type biomes due to its ability to withstand high environmental stresses. As migrating birds are suspected to have played an important role in facilitating the arrival of *A. cyclops*, further propagules are likely to reach the islands in the future. We recommend that new plantations of *A. cyclops* should be forbidden, but that extant naturalized populations should be managed instead of eradicating them. In fact, the effect of *Acacia* plantations warrants investigation at different scales in order to assess their impact on local plant-diversity and ecological succession processes.

**Keywords:** Alien plants, Mediterranean, Biological invasions, island vascular flora.

## Resumen

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Se muestran los dos primeros casos de naturalización de *Acacia cyclops* para el territorio italiano. Se observó que las plántulas de este árbol crecen en forma silvestre hace unos 15 años en Linosa (islas Pelagias, estrecho de Sicilia). Una década después, ya no puede ser considerada una xenófita casual ya que se observó una regeneración muy intensa en algunos proyectos de reforestación en Lampedusa, la isla más grande del mismo archipiélago. Publicaciones anteriores sugieren la necesidad de un monitoreo cuidadoso del proceso de invasión en curso, ya que *A. cyclops* ha expresado un comportamiento altamente invasivo en otros biomas de tipo mediterráneo, debido a su alta tolerancia al estrés ambiental. Dado que se sospecha que las aves migratorias han jugado un papel importante en la llegada de *A. cyclops*, es probable que en el futuro otros propágulos lleguen a las islas Pelagias. Se recomienda la prohibición de nuevas plantas, aunque también el manejo de poblaciones artificiales ya presentes en vez de su erradicación. El efecto de la forestación con *A. cyclops*, de hecho, merece ser investigado a diferentes escalas, a fin de aclarar su impacto tanto en la diversidad florística local como en los procesos de sucesión ecológica.

**Palabras clave:** Xenófitas, Mediterráneo, invasiones biológicas, flora vascular insular.

## INTRODUCTION

### *Plant invasion within the Mediterranean*

Biological invasions are widely recognized to be among the most critical factors affecting native ecosystems, by altering their structure and functioning, and are one of the main threats to biodiversity on a global scale (Chapin & al., 2000; Sala & al., 2000). Although the number of invasive alien species (IAS *sensu* Pyšek & al., 2004) represents only a small fraction of introduced species, their ecological and economic impacts are significant and long lasting. A number of factors make Mediterranean-type ecosystems (MTEs, i.e. Mediterranean Basin, California, Central Chile, the Cape Region of South Africa and S-SW Australia) particularly prone to invasions (Brunel, 2005), as has been shown dramatically in South Africa and in SW Australia (e.g. Henderson, 2001). Despite a significant and intense anthropic pressure over most of its territory, the Mediterranean Basin is still characterized by a striking species and community richness. Moreover, due to its remarkable environmental heterogeneity, the Mediterranean Basin hosts a particularly diversified flora concentrated in a relatively small area (Heywood, 1995), such that although representing only 1.6% of the earth's surface, about 25,000 plant species (i.e. approximately 10% of the World's vascular flora: Cowling & al., 1996) thrive there. The highest levels of

endemism have been recorded on islands (Heywood, 2011), where the ecological threat posed by alien invasive species seems to be higher than in adjacent mainland areas (Gimeno & al., 2006; Hulme & al., 2007). It has been suggested that island biota appear more vulnerable to invasions due to the presence of empty niches in unsaturated communities, the absence of key functional groups, and a generally lower competitive ability of the native species (Hulme, 2004). Indeed, more than 350 naturalized alien plants were found on only five of the major Mediterranean islands by Lloret & al. (2005), justifying Hulme & al. (2007), who pointed out that “the islands of the Mediterranean Basin probably represent some of the ecosystems globally most at risk from invasive species”. This worrisome situation has a historical explanation, because the Mediterranean Basin has a long history of trade links with many regions of the world, and many plants were introduced over the centuries, mainly for forestry, ornamental and agricultural purposes, thereby increasing the likely arrival of invasive species (Hulme & al., 2007; Lambdon & al., 2008a). During the last century, the rate of introduction of invasive species has dramatically increased, and a similar trend is expected to maintain over the next decades (Lambdon & al., 2008a). In addition, human pressure on Mediterranean environments has radically transformed the natural landscape and vegetation communities, particularly in coastal and upland

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areas (Gritti & al., 2006; Heywood, 2011). Land-use change (e.g. urbanization, exploitation of coastal areas, agricultural development, afforestation, etc.), together with the increased intensity and frequency of natural disturbance regimes, have markedly changed resource availability (light, water and nutrients), often favouring the establishment and the invasive success of allochthonous species (Lambdon & al., 2008b). As most of these species are strictly confined to frequently disturbed and anthropogenic sites, such as ruderal habitats, roadsides and waste places (Arianoutsou & al., 2010; Affre & al., 2010), many native ruderal annuals, a significant component of endemic Mediterranean plants (Heywood, 1995), might be seriously threatened by the spread of invasives. Among natural habitats, wetlands and coastal areas are relatively richer in IAS than other habitats, suggesting that those habitats may suffer a greater impact (Chytrý & al., 2009; Arianoutsou & al., 2010). In contrast, Mediterranean mature and stable woodlands, dominated by evergreen shrubs and trees, seem to be quite resistant to invasion (Affre & al., 2010; Arianoutsou & al., 2010;), although human disturbance, such as wildfires or silvicultural interventions, may alter canopy structure and so understory light availability, thus allowing the establishment of heliophilous tree species (Constán-Nava & al., 2010; Badalamenti & al., 2012); invasions by such species have already caused a reduction in species richness and a significant alteration of functioning of Mediterranean ecosystems (Vilà & al., 2006; Traveset & al., 2008). However, the major ecological impacts caused by IAS have probably yet to be realized, because a worsening of the current situation is expected, both in terms of number and spread, and conse-

quently of their likely damaging effects. Indeed, increasing anthropogenic pressure, together with the impact of climate change, could exacerbate the likelihood and the rate of spread of some IAS, thereby increasing habitat vulnerability or IAS competitive ability or both (Gritti & al., 2006; Affre & al., 2010). According to global scenarios, biodiversity in Mediterranean ecosystems could be threatened by IAS to a greater extent than in any other terrestrial biome during this century (Sala & al., 2000). The development of appropriate strategies in the Mediterranean Basin is urgently needed to tackle the IAS problem, which is increasingly becoming a priority in management policies of natural and semi-natural habitats.

### *The invasion of Acacia species in Western Mediterranean*

With regard to the genus *Acacia*, at least 23 species figure among the most aggressive invaders of Mediterranean-type ecosystems worldwide, and 14 of them (Table 1) have been particularly successful in colonizing southern European countries (Lambdon & al., 2008b).

After successful establishment, several wattle species have proved capable of altering the community structure (Marchante & al., 2008; Werner & al., 2008), not only because of their marked ability to sequester water resources (Rascher & al., 2011), but also because their nitrogen fixing activity (Hellmann & al., 2011), and the release of allelopathic compounds (Hussain & al., 2011) that completely alter the biochemical and microbiological characteristics of the soils of the invaded ecosystems.

As a consequence, special efforts have been made to fore-

**Table 1.** An overview of the 14 recorded as invasive *Acacia* species within Mediterranean countries and their: present status in Sicily.

Taxon	Origin	Record of cultivation	Date of naturalization	S&a	T&B	ILDIS	CG	R&R
<i>A. baileyana</i> F. Müller	SE Australia	1910 (Ostinelli)	n.d.a.	+	—	—	—	—
<i>A. cyclops</i> G. Don	SW Australia	c. 1910 (Borzi & al., 1911)	2012 (this paper)	+	inv	—	—	—
<i>A. dealbata</i> Link	SE Australia & Tasmania	1903 (Zodda)	n.d.a.	—	inv	—	inv	+
<i>A. decurrens</i> Willd.	SE Australia	1821 (Gussone)	n.d.a.	+	—	—	—	—
<i>A. farnesiana</i> (L.) Willd.	Santo Domingo	XVII century	1968 (Franco)	+	—	+	cas	—
<i>A. karroo</i> Hayne	S Africa	1821 (Gussone, sub <i>A. horrida</i> )	1957 (Romeo, sub <i>A. horrida</i> )	+	inv	+	nat	—
<i>A. longifolia</i> (Andrews) Willd.	SE Australia	1821 (Gussone)	1909 (Trinchieri)	+	inv	—	nat	+
<i>A. mearnsii</i> De Wild.	SE Australia & Tasmania	1993 (Paiero & al.)	n.d.a.	+	inv	—	inv	+
<i>A. melanoxylon</i> R. Br.	SE Australia & Tasmania	1903 (Zodda)	1909 (Trinchieri)	+	inv	—	cas	+
<i>A. pycnantha</i> Benth.	SE Australia	c. 1910 (Borzi & al., 1911)	1977 (Lumini, 1978)	+	inv	—	cas	—
<i>A. retinodes</i> Schldl.	SE Australia & Tasmania	1910 (Ostinelli)	n.d.a.	+	inv	—	inv	—
<i>A. saligna</i> (Labill.) H.L. Wendl.	SW Australia	1903 (Zodda, sub <i>A. cyanophylla</i> Lindley)	1968 (Franco, sub <i>A. cyanophylla</i> Lindley)	+	inv	+	inv	—
<i>A. sophorae</i> (Labill.) R. Br.	SE Australia	n.d.a.	n.d.a.	+	—	—	—	—
<i>A. verticillata</i> (L'Hér.) Willd.	Australia & Tasmania	1821 (Gussone)	n.d.a.	+	—	—	—	—

S&a, Naturalized in S Europe according to Sheppard & al. (2006); T&B, species behaving as invasive weeds in at least one Mediterranean country according to Thomson & Brundu (2010); ILDIS, presence in Sicily according to ILDIS (2010); CG, mentioned in the most recent check-list of Italian alien plants (Celesti-Grapow & al., 2010); R&R, widespread xenophytes showing weedy behaviour in Europe according to Richardson & Rejmánek (2011); cas, casual; nat, naturalized; inv, invasive; n.d.a., no data available.

cast or avoid the risks of invasion of the most dangerous *Acacia* species, such as *A. mearnsii* (Lowe & al., 2000), *A. longifolia* (Marchante & al., 2008), *A. saligna* (Crosti & al., 2010) and *A. dealbata* (Lorenzo & al., 2010).

### Global distribution and ecology of *Acacia cyclops*

Easy to recognize by the conspicuous dark orange-red eye-mimicking aril surrounding the seed (Kodela & Tindale, 2001), within its native range *Acacia cyclops* grows along the coasts of W and S Australia, which are mostly characterized by a Mediterranean-type climate (Gill, 1985). Widely cultivated in many warm-temperate and subtropical regions of the world, *Acacia cyclops* has also been used for afforestation in semiarid regions because of its drought and salt tolerance (National Research Council, 1980), and it is able to establish in areas with only 250–300 mm of annual rainfall (National Research Council, 1980; Gill, 1985). It has been introduced as an ornamental plant or for afforestation purposes in the Mediterranean Basin, where it is naturalized in northern Africa (Vilà & al., 1999), along the coasts of S Spain (Paiva, 1999) and W Portugal, where it behaves as an invasive environmental weed (Marchante & al., 2008). In recent years it has also achieved a very rapid spread in the Canary Islands (Verloove & Reyes-Betancort, 2011). In Italy, *Acacia cyclops* – commonly named red-eyed wattle – does not figure in the most recent alien plant inventories (e.g. Celesti-Grapow & al., 2010).

Currently, *A. cyclops* is the most widespread and dangerous woody invader in the Cape Province, South Africa (Whibley & Symon, 1992; Richardson & al., 1996), where it may form dense, impenetrable stands invading and displacing indigenous flora. Once established over large areas, the species is difficult to remove or replace, especially after a consistent seed bank in the soil has been formed (Milton & Hall, 1981). Attempts to control it have included the use of both herbicides and biocontrol agents (Henderson, 2001), as well as prescribed burning, alone or combined with felling (Holmes, 1989). However, two insects have recently provided a significant reduction in *Acacia cyclops* seed production (Impson & al., 2004; Post & al., 2010), and biological control appears to be the most effective and cost-saving method for curbing the Acacias reproductive potential and related invasiveness (Richardson & Kluge, 2008). Because of the substantial absence of vegetative propagation (Gill, 1985; Gibson & al., 2011), *A. cyclops* establishment and invasive success is strongly dependant on a copious production of viable seeds and their subsequent dispersal (Milton & Hall, 1981; Holmes, 1988). In particular, its seeds with attractive arils are well adapted for bird dispersal whilst they are retained in the canopy (Glyphis & al., 1981), but once the seeds have fallen to the ground, ants and small rodents may also spread them or carry them into the deeper soil layers, thus increasing the likelihood of their being added to the soil seed-bank (Holmes, 1989; Richardson & Kluge, 2008).

Unlike other well-known invasive Acacias, such as *Acacia saligna* (Labill.) H.L. Wendl. and *A. longifolia* (Andrews) Willd., *A. cyclops* seed germination does not seem to be significantly enhanced by very high temperatures or by fire (Jef-

fery & al., 1988; Holmes, 1989), rather it depends upon chemical scarification, notably after the passage through the gut of birds, or to only moderate heating in order to break dormancy (Glyphis & al., 1981; Gill, 1985; Jeffery & al., 1988). Its known ability to regenerate prolifically following fire from the huge seed-bank stored in the soil (Milton & Hall, 1981), and its dominance over frequently burned areas in South Africa (Glyphis & al., 1981; Richardson & al., 1996), is therefore mainly due to the reduction of interspecific competition caused by fire rather than to a higher seedling emergence and establishment. Moreover, the considerable annual seed rain (nearly 2,000 seeds/m<sup>2</sup>), together with the high percentage of seed physical dormancy (Milton & Hall, 1981), indicate that this species rapidly accumulates and stores its seeds in the soil. But since only 20–40 % of seeds remain dormant in the soil after one year (Holmes & Moll, 1980), due to germination or decay, the time to produce an effective soil seed-bank takes 11 years (Holmes, 1989). From that moment on, the seed density in the soil stabilizes, and may range from 1,400 to 7,800 seeds/m<sup>2</sup> (Holmes & al., 1987; Holmes, 1989).

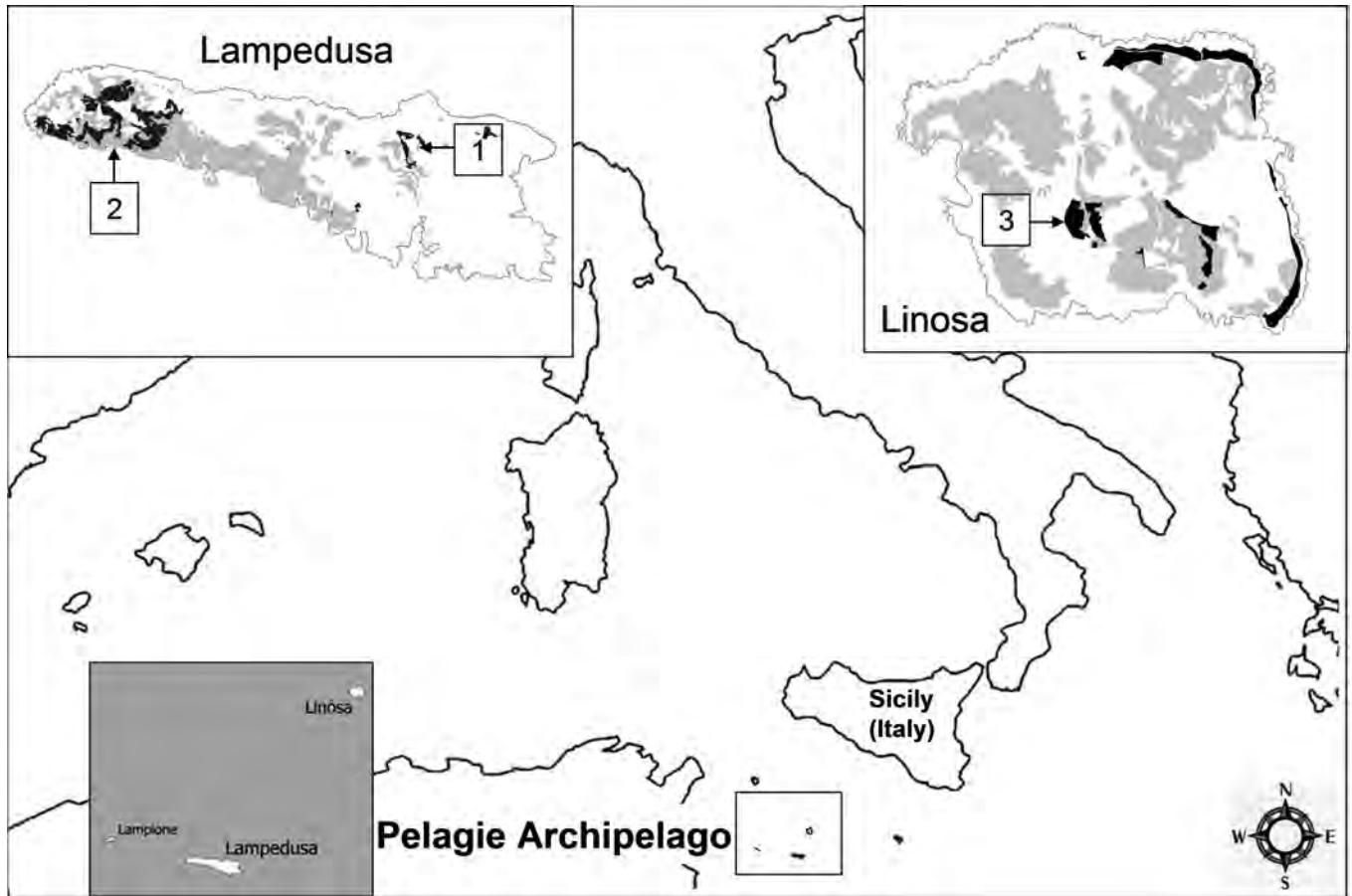
The South African invasion case clearly demonstrates that an *A. cyclops*-dominated community is persistent (Richardson & Cowling, 1992) and that this species has a great ability to expand out of its native range (Higgins & al., 2000). In addition, its definitive establishment results in constant, large inputs of nitrogen-rich litter because this xenophyte produces more litter with a higher leaf N concentration than native fynbos shrubs, increasing not only total soil N and organic matter (Witkowski, 1991a), but also net N mineralization rates (Stock & al., 1995). In many nutrient-limited ecosystems this evergreen shrub shows a great competitive ability, a higher relative growth rate (Witkowski, 1991b); furthermore, a better acquisition of water resources (Morris & al., 2011) allows it to out-compete native coexisting species.

## MATERIAL AND METHODS

In order to provide a preliminary evaluation of *A. cyclops* reproductive success, we measured the percentage cover of seedlings (bearing the juvenile character of compound leaves) and saplings (in which the compound leaves have been replaced by phyllodes) under twenty mature plants at Taccio Vecchio, where an intense self-regeneration rate had been observed. The same plants were used to check the effect of red-eyed wattle on the recruitment and establishment of native woody species. We also carried out phytosociological relevés (Braun-Blanquet, 1932) in two 5 m<sup>2</sup>-wide circular concentric areas around the stem, the first under tree canopy (u), the second outside the canopy (o), i.e. in open areas adjacent to the investigated plants. In order to test any facilitating effect of *A. cyclops*, the total woody (subshrubs, shrubs and trees) species richness, and plant and litter percentage cover were recorded, and also the number of species typical of garrigue, maquis and forest communities, except *A. cyclops*.

## RESULTS AND DISCUSSION

Our report represents the first case-study of naturalization of *Acacia cyclops* in Italy. On Lampedusa (Strait of Sicily, Pelagie Islands), interviews with local forest agents, land own-



**Fig. 1.** Location of Pelagie Archipelago, Lampedusa (Strait of Sicily, Southern Italy), and *Acacia cyclops* study areas (boxes: 1, Taccio Vecchio; 2, Sanguedolce, Lampedusa. 3, Monte Bandiera, Linosa). The black areas represent the distribution of the afforestations while the light-grey ones represent the distribution of garrigue communities.

ers and shepherds allowed us to retrace the intriguing history of the local naturalization of *A. cyclops*. The first red-eyed wattle was observed growing wild in the 70s near a private orchard at Contrada Sanguedolce. Since no one had ever seen the plant before, and the owner testified that he had not planted it (R. Almanzo, *pers. comm.*) seeds were probably introduced by birds migrating from Tunisia. In following years, the forest agent Calogero Mantis collected seeds from the first settler plants to become established, sowed them in the nursery on Lampedusa, and sent some young plants to Linosa (Mantis, 2001). Subsequently, in August 1998, *A. cyclops* was observed to produce seeds and had become established within a private camping area on Linosa (Strait of Sicily, Pelagie Islands) by S. Pasta and S. Mattana. A decade later (June 2009 according to T. La Mantia and S. Pasta), seedlings of *A. cyclops* were also observed growing within the afforested areas on Lampedusa, the major island of the same Archipelago (Fig. 1).

Intense regeneration by self-sowing has been observed (Fig. 2) in Contrada Sanguedolce (35°31'19"N, 12°32'35"E, 97 m a.s.l.), and subsequently (March, 2012) in Contrada Taccio Vecchio (35°31'02"N, 12°36'06"E, 46 m a.s.l.).

In recent years a significant increase of the number of seedlings, saplings and young trees has been recorded both on Lampedusa and Linosa; Effectively, the area colonized by

*A. cyclops* is about 6 ha on Lampedusa (5 ha in Contrada Taccio Vecchio + 1 ha in Contrada Sanguedolce) and about 0.3 ha at Linosa. Although the most intense renewal and spread of the invading species has been recorded within afforested



**Fig. 2.** Lampedusa, Locality Taccio Vecchio: vigorous germination under *Acacia cyclops* canopy.

areas, the first attempts to spread out of the artificial plantation limits were observed since 2008. This success may be enhanced by local climate, which matches very well *A. cyclops* requirements: local mean yearly temperature is about 19 °C, while the mean annual rainfall, mostly concentrated between October and March, has decreased from 500 to about 300 mm in the last 150 years (La Mantia & al., 2011).

Vegetation relevés (Table 2) reveal that total and subshrub species richness is higher within open habitats. On the other hand, all the other selected indicators (i.e. number of woody taxa, plant coverage, number of taxa referred to 'final' stages of succession) suggest a facilitation effect of *A. cyclops* plantations. In effect, all the seedlings and saplings of *A. cyclops* and of native woody species were found under adult *A. cyclops* trees.

Current investigations aim at understanding how planted *A. cyclops* improves renovation, in order to assess the role of different factors such as 1) increase of soil humidity due to litter coverage, 2) milder microclimatic conditions due to canopy shading, 3) nutrient input due to root symbiotic microorganisms and 4) nutrient input due to litter decomposition.

## CONCLUSIONS

Owing to its strong drought tolerance, we assume that the expected reduction of rainfall in future decades will not hamper the local expansion of *A. cyclops*. On the contrary, more severe stresses may improve its competitive ability against native woody species. In fact, despite the presence of natural regeneration of *Thymbra capitata* (L.) Cav., *Thymelaea hirsuta* (L.) Endl., *Pistacia lentiscus* L., *Prasium majus* L., *Asparagus aphyllus* L. and *Ceratonia siliqua* L. (Fig.3), the overwhelming number of *A. cyclops* saplings, growing as clumps under the canopies of mother plants, suggests that facilitative conditions are much more favourable for self-regeneration than for the establishment of other co-existing species. Therefore, this alien plant may prevail over the native species listed above, at least in a long-term perspective. In fact, if we consider that Lampedusa and Linosa host many birds (Table 3) that are already known to play a role in the dispersal of *A. cyclops* in



**Fig. 3.** Lampedusa, Locality Taccio Vecchio: saplings of *Pistacia lentiscus* under *Acacia cyclops* canopy.

South Africa (e.g. *Streptopelia senegalensis* and *Sturnus vulgaris*: Glyphis & al., 1981), both these islets are likely to be prone to repeated introductions from N Africa, and to its spread in new suitable localities.

Recent investigations carried out on Lampedusa underline that the establishment of native woody species is a very difficult and slow process both in seminatural open habitats such as grasslands and garrigues (Pasta & al., 2007) and in *Pinus halepensis* Mill. plantations (Pasta & al., 2012). As a consequence, local plantations of short-lived Acacias were made in order to obtain ecological services (i.e. plant cover increase, soil protection and fertilization, microclimate amelioration, etc.) and to accelerate local succession, rather than to obtain wood or fodder. In this sense they have worked rather well, especially in the case of *Acacia saligna*, which up to now has not given any evidence of naturalization on Pelagie islands (Pasta & La Mantia, 2001).

Notwithstanding the above-mentioned habitat modification provided by arboreal Leguminosae, and although up to now *A. cyclops* naturalization seems to be largely limited to forest plantations, a constant monitoring of its spread on Lampedusa and Linosa is needed because of its known invasiveness worldwide and because the Pelagie Islands host a very precious botanical heritage, with some 20 endemic and nearly one hundred other rare or threatened plants (La Mantia & al., 2009).

Considering both the pros and cons of *A. cyclops* in Mediterranean environments, in our opinion the best way to deal with it is neither to directly favour introductions (planting new individuals should be forbidden) nor to eradicate its local populations, but to control and to manage them through selective cutting where the native shrub vegetation has developed. During last four decades Sicily experienced a dramatic rise of successful invasion events by other woody Mimosoideae, such as *Parkinsonia aculeata* L. (Orlando & Grisafi, 1977), *Paraserianthes lophantha* (Willd.) I.C. Nielsen (Villari & Zaccane, 1999), *Albizia julibrissin* (Willd.) Durazz. (Gueli & Lo Giudice, 2005), *Senegalia visco* (Griseb.) Seigler & Ebinger and *Leucaena leucocephala* (Lam.) De Wit

**Table 2.** Vegetation characterization in studied populations. Life-forms according to Raunkiaer (1934).

Parameter	under the canopy	outside the canopy
<i>A. cyclops</i> seedlings coverage (%)	13	0
<i>A. cyclops</i> saplings coverage (%)	2.4	0
Species richness (= tot. nr. of taxa)	10.4 ± 6.0	12.8 ± 2.0
Nr. of woody taxa (P + NP + Ch)	6.0 ± 3.0	3.6 ± 0.5
Plant coverage (%)	33.0 ± 18.0	20.0 ± 12.0
Litter coverage (%)	64.0 ± 28.0	0
Nr. of taxa typical to maquis and forest communities (Q)	2.6 ± 2.0	0
Nr. of taxa typical to garrigue communities (C-M)	2.2 ± 1.0	3.0 ± 1.0

P, Phanerophyta; NP, Nano-Phanerophyta; Ch, Chamaephyta. Phytosociological classes according to Rivas-Martínez & al. (2001): Q, *Quercetea ilicis* Br.-Bl. ex A. & O. de Bolòs 1947; C-M, *Cisto-Micromerietea julianae* Oberdorfer 1954.

**Table 3.** A comparison between the South African birds which act as *Acacia cyclops* seed dispersers and their potential substitutes on the Pelagian islands.

S Africa	Lampedusa and Linosa
<i>Passer melanura</i> (Müller, 1776)	<i>Passer hispaniolensis</i> Temminck 1820
<i>Serinus sulphuratus</i> Linnaeus 1766	<i>Serinus serinus</i> Linnaeus 1766 <i>Carduelis cannabina</i> Linnaeus 1758
<i>Streptopelia capicola</i> (Sundevall, 1857)	<i>Streptopelia decaocto</i> Frivaldszky 1838
<i>Streptopelia senegalensis</i> (Linnaeus, 1766)	<i>Streptopelia senegalensis</i> (Linnaeus, 1766)
<i>Sturnus vulgaris</i> Linnaeus 1758	<i>Sturnus vulgaris</i> Linnaeus 1758

subsp. *glabrata* (Rose) S. Zárate (Raimondo & Domina, 2007).

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