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Plant invasions in Mediterranean island ecosystems.
Impacts to biodiversity and risk of invasion in the Tuscan Archipelago: from ecology to management.

Tesi di
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ABSTRACT

The researches within this PhD thesis focus on the problem of non-native plants in the Tuscan Archipelago, particularly investigating distributions, invasiveness, impacts on diversity of alien plants and management options for the invasive alien plants. The PhD thesis is here presented as a collection of the resulting contributions, including poster presented at international conferences and papers published or in preparation. The investigations on the distributions and invasive status of alien species in the Tuscan Archipelago led to the preparation of a checklist of the alien flora of the Tuscan Archipelago. The ecological traits of plant invasions in the Tuscan Archipelago were explored towards the research on potential impacts due to invasive species on the Island of Elba. These studies allowed us to detect the impacts on plant richness and diversity, soil chemical properties and soil microbial communities of *Acacia dealbata* and impacts on plant communities of *Acacia pycnantha*. Finally, regarding management aspects, the work dealt with the evaluation of the risk of invasion through two different approaches. The invasiveness of alien species in the Tuscan Archipelago was assessed by use of risk assessment procedures. We assessed the risk of invasion related to 212 alien plants in Tuscan Archipelago testing two different procedures of Weed Risk Assessment. The risk of invasion by alien species on habitats worthy of conservation was investigated adopting Species Distribution Models. The potential distributions of six harmful invasive plants in Mediterranean Ecosystems were merged with the density of habitat win the Island of Elba to obtain a MAP of Risk of invasion. Thanks to the exploration of these topics we highlighted different aspects of the biology of invasions in this representative Mediterranean island ecosystem. We provided both theoretical contributions to the issues on the impacts of invasive species and technical tools for conservation, such as monitoring of the current distributions and evaluation and prioritization of risk of invasion.

1. INTRODUCTION

Biological invasions

Since the publications in 1958 of Elton's work *The ecology of invasions by animals and plants* (Elton 1958) the study of biological invasion has gained an increasing importance in the global scientific and nature conservation framework. In deed in the last few decades, a plethora of studies has focused on several aspects of this topic, ranging from theoretical and ecological features to the management options. This exploration includes the continuous research of a unified terminology background, the depiction of the ecological processes related to species introduction and spreading, the practical search through impacts and threats related to species invasion and the actions that have to be undertaken to mitigate the potential impacts of biological invasion on biodiversity and human activities. Nowadays invasive alien species and impacts related to biological invasions are considered one of the major threats to biodiversity worldwide (CBD decision VI/23, 2014).

Even though, or probably due to this global interest in this framework still some confusion is present among the correct definitions. A quite good agreement can be found for the alien definition. According to Richardson et al. (2011) alien species, often referred also as exotic, introduced, non-indigenous, or non-native species, are those species moved by human activities beyond the limits of their native geographic range into an area in which they do not naturally occur. The movement allows the species to overcome fundamental biogeographic barriers to their natural dispersal. This definition is quite coherent with those adopted by the World Conservation Union (IUCN)

> "a species, subspecies, or lower taxon occurring outside of its natural range (past or present) and dispersal potential (i.e. outside the range it occupies naturally or could not occupy without direct or indirect introduction or care by humans) and includes any part, gametes or propagule of such species that might survive and subsequently reproduce"

And Convention on Biological Diversity (CBD)

> "A species, subspecies or lower taxon, introduced outside its natural past or present distribution; includes any part, gametes, seeds, eggs, or propagules of such species that might survive and subsequently reproduce."

The focal point, generally agreed in the above definitions of "alien species" is the human related movement, associated to both unintentional or intentional introduction, that allows the species to overcome geographical barriers (Richardson et al. 2000; Pyšek et al. 2004; Blackburn et al. 2011; Blackburn et al. 2014). Regrettably, the same agreement cannot be found on the individuation of "invasive alien species". According Richardson et al. (2011), and several other previous works (see again Richardson et al. 2000; Pyšek et al. 2004) the status of invasive regards mainly the capability of an alien species to produce self-replacing population and produce reproductive offspring, leading to a conspicuous spread in space in the introduced range. As precisely stated this definition explicitly excludes any connotation of impact and is based exclusively on ecological and biographical criteria. On the other hand, it has to be noted that impacts are widely considered as an important part of the definition of "invasive species", as done in the definitions adopted by IUCN, where an invasive alien species is defined as

> "an alien species which becomes established in natural or semi-natural ecosystems or habitat, is an agent of change, and threatens native biological diversity."

Or by CBD:

> "an alien species whose introduction and/or spread threaten biological diversity"

Impacts are indeed a quite important topic in invasion biology, and an essential component in the management of biological invasion. It is, of course, important to acknowledge that many non-native species introduced outside their natural territories do not represent a threat in the new range (Williamson 1997) and that generally invasive and problematic species do represent a narrow subset of alien species. Furthermore many introduced species are an important resource for local and national economies (van der Weijden et al. 2007). Nevertheless, impacts related to IAS represent nowadays a focal challenge for policy and managements purposes.

Particularly focusing on plants, the impact of invasive (or more generally, alien) plants on invaded ecosystem can be various and have been largely studied with several approaches (Pyšek et al. 2012; Jeschke et al. 2014). Alien species produce an impact since they determine any changes in the ecosystem in which they have been introduced. Impacts represent generally the description or quantification of how an alien species affects the physical, chemical and biological environment (Richardson et al. 2011). Alien species can have detrimental effects on various elements of the invaded area, generating ecological and socioeconomic impacts. Incidentally, it should be probably noted here that all these compartment should be ideally summarized in the term *ecosystem*.

Ecological impacts on native species, communities and ecosystems have been largely studied for many invasive alien plants. IAS are often related to decrease in species richness and diversity, and to the reduction of distinctiveness of local biologic communities. According to Vilà et al. (2014) it has been globally demonstrate that at a local scale non-native plants can cause a decrease in plant and animal species richness, also resulting in a series of cascade effect with detrimental effect on the invaded ecosystems. Impacts on species and ecosystems include also genetic variation via hybridization with native population (Vilà et al. 2000). Many invasive alien species can alter the functionality of invaded ecosystems, leading to changes in the communities structure and species assemblages. For instance, N-fixing plants can drive a process of nitrification affecting several compartments of the ecosystem, going from soil chemical properties to soil microbial, understory plant and lichens communities. Impacts of alien species can also lead to the *biotic homogenization* of invaded ecosystem, defined as the increase in biological similarity between communities through time (Mc-Kinney & Lockwood 1999, Olden & Poff 2004). The process of biotic homogenization can affect biodiversity at several levels: taxonomical, genetical or functional (Olden & Rooney 2006), and across different geographical scales and time frames (Ricotta et al. 2012; Campos et al. 2013).

Biological invasion also cause important economic impacts, related to direct and indirect financial costs. Direct costs are linked primarily to nature conservation, agriculture, fisheries and forestry as main economic sectors affected by IAS. A crude estimation of total known monetary cost of alien species in Europe is close to €10 billion per year (Hulme et al. 2009). Kettunen et al. (2009), in the technical support to EU strategy on IAS assessed the total documented monetary impacts of IAS in Europe to a total of €12.5 billion per year, summing €9.6 billion resulting from the damage caused by IAS and €2.8 billion related to the control of IAS. Across control costs, those related to terrestrial IAS (e.g. vertebrates, plants and invertebrates) form a major part of this estimate. While across documented costs those related to terrestrial plant amount to about €3.7 billion. The estimation of total costs of IAS in Europe arise to €20 billion per year, which as the authors themselves acknowledge still represent an underestimation of real costs.

Concluding this short summary of impacts related to biological invasion it should be noted, as it has been frequently pointed out, that ecological and socio/economic dimension of the problem of invasive species are actually connected at different levels. The ecological changes that lead to a lower resilience of ecosystems to invasion and related impacts are highly correlated to the growth of transport, trades and market globalization that enhance the high movement of species across geographical barriers. Another essential link is represented by the loss of functionality and productivity by invaded ecosystem that are subsequently related to depletion in the ecosystem services and economic losses.

Plant invasion in Mediterranean islands ecosystems

It is generally agreed that effect of biological invasion are going to be more dramatic on insular ecosystem (Hulme 2004) and that islands are more threatened by plant invasion than mainland. Islands host peculiar biomes, usually poor and disharmonious in species and rich of endemics (Whittaker 1998; Whittaker and Fernández-Palacios 2007), species on islands have low vagility and usually form small populations which existence is susceptible at various level from IAS (Berglund et al. 2009). Furthermore insular ecosystems host often more alien species compared to mainland (Lonsdale 1999), and their biomes appear more disturbed and dominated by alien species (D’Antonio & Dudley 1994). Pysek et al. (2012) underlined an important role of insularity and Mediterranean biome as drivers of impacts of IAS on native richness. For example Vilà et al. (2014), in a meta-analysis on the literature on impacts of alien plant, showed that the impact on animal richness, mainly arthropods, is generally stronger on islands. On the other hand it is noteworthy that impacts of invasive alien plants on native plant richness appeared not significantly greater in island that in mainland (Vilà et al. 2011, 2014). However it has been suggested that ecological impacts of plant invasions on islands can be more related to changes in species assemblages, and in replacement of endemic species than in a net decrease in the number of species (Vilà et al. 2014).

Mediterranean basin, with its complex system of archipelagos, islands and islets, represent an important area of species diversity. It is usually reported that Mediterranean basin vascular flora amount at about 24000-25000 species, accounting for 10% of world plant richness, with at least 13000 endemics (Brundu 2013). This has led to recognition of the Mediterranean basin as one of the 34 Global Biodiversity Hotspots (Mittermeier et al. 2004). On Mediterranean islands, the rates of endemism often exceed 10%, and sometimes 20%, of local flora (Medail and Quezel 1999). Furthermore, this endemism is characterized by an extremely high rate of narrow endemism, reaching the 60% of the total. Island represent indeed a quite important trait in the Mediterranean basin. Although a comprehensive assessment of the real amount of island for the Mediterranean basin is extremely difficult, a total of 10000-15000 islands appear a reasonable estimate for this number (Brundu 2013).

According to the future scenario proposed by Sala et al. (2000), Mediterranean ecosystems may be prone to an increase in the invasion rates. Especially Mediterranean islands are particularly vulnerable to biological invasion due to their peculiar biomes (Hulme et al. 2007). Furthermore the effects of plant invasions in Mediterranean islands appear enhanced as the result of changes in important driving factors, e.g. disturbance regimens, land use and climate (Pretto et al. 2010, 2012). Indeed in insular ecosystems, even more than in mainland, human related factors play an essential role in the raising of risk related to biological invasion. In the last decades Mediterranean islands were generally interested by deep changes in their socio-economic status, with the transition from an economy based primarily on agriculture and livestock farming to one based essentially on mass tourism (Delanoë et al. 1996). This leaded to the one hand to a massive increase in alien species introduction and on the other hand to an increase in urbanization and disturbance that lower the resilience of this ecosystems to invasion, with the creation of dramatic synergies related to human activity.

Concluding this paragraph, as stated in the CBD report (CBD decision VI/23 2014),

> “invasive alien species represent one of the primary threats to biodiversity, especially in geographically and evolutionarily isolated ecosystems, such as small island developing States, and that risks may be increasing due to increased global trade, transport, tourism and climate change.”

Thus, the study of biological invasion (in our specific case of plant invasion) is a central pillar for future needs of nature conservation on islands.

A particular need for developing basic knowledge, impact information and risk assessment on islands ecosystems.

In spite of a certain interest in the study of invasions in Mediterranean ecosystems (see Gaertner et al., 2009 and reference therein) and the general recognition of plant invasion as a major threat in these ecosystems (Underwood et al., 2009; CBD decision VI/23, 2014), information on plant invasion in Mediterranean island ecosystems are still lacunose and discontinue, especially on Protected Areas in Mediterranean Islands (Brundu, 2013). These information gaps regards several issues, going from the lack of a precise knowledge of the presence of alien plants to the understanding of the ecological impacts related to plant invasion. In addition, there is an urgent need of the definition of priority and the adoption of specific strategies and policies (Brundu 2013).

Within this framework, aim of this work is to provide a comprehensive study of plant invasion in the Tuscan Archipelago. This three years research is conceived to represent a contribute on the knowledge on this subject, encompassing several important topics. We mainly aimed to

i) develop a solid and comprehensive basic knowledge on the alien flora of the Tuscan Archipelago, indicating distributions and invasion statuses of the alien plants across the island of the Archipelago;

ii) investigate the ecological consequences of the process of invasion studying, as a particularly significant example, the effects on the ecosystems of two highly invasive species such as *Acacia dealbata* and *Acacia pycnantha*;

iii) apply the theoretical and ecological knowledge aiming to assess the risk of invasion for the alien species present in the Tuscan Archipelago and to asses which habitat are more at risk and where.

We presented this thesis as a collection of all the contribution we produced according the aim of understand the invasion process in an archipelago and its consequences on the local biota. This collection includes, after a brief introduction on the main sections, some minor contribution, such as paper in Italian for national journals and poster presented at international congresses, and five main contributions, already published, submitted or in preparation for international journals. We followed an ideal route, starting from the improvement of our knowledge and comprehension of the phenomenon in the Tuscan Archipelago, to finally develop risk analysis and risk management procedures throughout an optimization of management options. So firstly, we produced an updated checklist of the alien flora of the Tuscan Archipelago (Sec 3.3). Regarding ecological impacts, we produced two main contributions. A first paper investigate the impacts of A. dealbata on soils chemical properties, soil microbial communities and understory plant communities (Sec 4.2). A second work regards impacts of A. pycnantha on understory plant communities (Sec 4.3). Finally aiming to assess the risk of invasion, we analyze the problem firstly by testing two risk assessment procedures on the alien species of the Tuscan Archipelago (Sec 5.3). Thus, we used habitat suitability models to assess potential presence of six particularly harmful IAS and verify where these species could invade valuable habitats (Sec 5.4).

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2. AREA OF STUDY: THE TUSCAN ARCHIPELAGO

The researches object of this PhD thesis focus on the Tuscan Archipelago territories. The Tuscan Archipelago (Fig 2.1) is located in the central Mediterranean Sea, west of Tuscany (Italy) and east of Corse. It extends through a sub-triangular area, with extremes between 42°13'42" N (Giannutri) and 43°26'54" N (Gorgona) and 9°43'18' E (Capraia) and 11°11'00" E (Giannutri).

It consists of seven main islands: Gorgona, Capraia, Elba, Pianosa, Montecristo, Giglio and Giannutri (from North to South) plus several minor islands and islets, large parts of which are included in the Tuscan Archipelago National Park (TANP). The area under protection by TANP include also 567 Km^2 of sea.

Figure 2.1 Geographical overview of the Tuscan Archipelago.

The islands of the Archipelago account for a total surface of around 300 square kilometers. The Island of Elba, the largest island of TANP, has a surface of 224 Km^2, while the others islands are quite smaller, with surfaces ranging from 20 to 2 Km^2 (Table 2.1). Resident population is mainly restricted to the islands of Elba (around 33000 inhabitants) and Giglio (with approximately 1500 inhabitants), while the other islands host from some hundreds to few units of people. Montecristo is the less populated island with only two residents.

A recent review of geological, geomorphological and climatic traits of the Tuscan Archipelago can be found on D'Orefice and Foresi (2009). The quite different geologic substrata that can be found among the islands highlight the complex geologic history that characterized its formation. The Island of Gorgona is formed by metamorphic rocks. Capraia is a volcanic Island. Montecristo is entirely granitic, as is almost the entire Island of Giglio. Pianosa consists of sedimentary rocks and shell formations, while Giannutri is formed entirely by dolomitic-limestone sediments, which have much in common with the nearby Apennine Chain. The Island of Elba is highly heterogeneous, with a prevalence of granites in the western part and of metamorphic rocks on the eastern part of the Island (D'Orefice and Foresi 2009).

In Table 2.1 are summarized the most important geographical information.

Island	**Surface (Ha)**	**Altitude (m)**	**Substrate**	**Number of spontaneous plants**	**Pop**
Capraia	19.3	447	trachytes	600 (Foggi & al. 2001; unpubl.)	410
Elba	224.4	1018	granites, trachytes, limestone, metamorphic	1250 (Fossi Innamorati 1983, 1989, 1991, 1994, 1997; unpubl.)	33000
Giannutri	2.4	93	limestone/panchina	350 (Baldini 2001; unpubl.)	15
Giglio	21.5	498	granites/limestome	700 (Baldini 1998; unpubl.)	1500
Gorgona	2.3	255	metamorphic	500 (Rizzotto 2011; unpubl.)	150
Montecristo	10.4	645	granites	400 (Paoli & Romagnoli 1976; unpubl.)	2
Pianosa	10.3	27	limestone/panchina	550 (Baldini 2000; unpubl.)	10

Table 2.1 Geographical information on the seven main islands of the Tuscan Archipelago. Resident population is approximated after ISTAT 2011 data. Numbers of spontaneous flora are approximated according to recent unpublished data.

The islands differs among them greatly also regarding geomorphology. The Elba Island is quite heterogeneous, with rocky cliffs on eastern part, lowlands in the central part and the massif of Mount Capanne, reaching the height of 1018 meters above sea level, in the western part. Capraia, Montecristo, Giglio and Gorgona, still not reaching the altitude of M. Capanne, are characterized by generally acclive rocky cliffs and slopes (see Table 2.1 for altitudes). Finally Pianosa and Giannutri are entirely flat islands (D'Orefice and Foresi, 2009).

The climate of Tuscan Archipelago belong to typical Mediterranean climate, with the colder semester interested by western precipitation and the warmest month dominated by a dry and stable climate due to the anticyclones of Azores and Sahara. The temperature are generally high, with annual mean temperature ranging from 16.9°C for Pianosa to 13.8°C on Mount Calamita, while the lowest temperatures are registered on the top of Mount Capanne. The mean temperatures of the coldest month (January) range from 6°C (Elba) to 10°C (Pianosa). Mean temperatures of warmest month range between 22°C and 26°C. Precipitation regimes are a crucial factor in shaping the climate features of the Archipelago. As characteristic of Mediterranean and sub-Mediterranean climates precipitation are scarce and concentrated on autumn or winter. Precipitation in spring arise to 25% of the total and summer present a quite low minimum, even less than 10% of total annual precipitation (Aringoldi et al., 2009). This determine the classic summer drought that characterize Mediterranean climates and constraint Mediterranean biomes. According Thornthwaite & Mather (1957) classification the net prevalence of evapotranspiration on precipitation indicate the presence of sub-humid (C2), subarid (D) and arid (E) climates. Thus especially in the summer the availability of water in the soil is very poor.

The Archipelago hosts an extraordinary biota, although less rich than on the adjacent continental mass, particularly rich in endemic species, mainly in plants and invertebrates.

The most representative endemic animal species can be found among gastropod mollusks (*Oxychilus sp. pl.*) and insects. Other important species can be found among amphibian (*Hyla sarda* and *Discoglossus sardus*) and reptiles, with several endemic lizards (i.e. three endemic subspecies of *Podarcis muralis*). The land mammals are those typical to the Mediterranean environment. The pine marten (*Martes martes*), rare anywhere else, has a strong presence and is an agile inhabitant of the woods of Elba. However a strong presence of introduced mammals can be found across the islands, such us introduced mouflons (*Ovis aries*), feral goats (*Capra hircus*), central European wild boars (*Sus scrofa*) and European hares (*Lepus europaeus*). The presence of birds is particularly enriched by the occurrence of migratory species. The Tuscan Islands are indeed an important migratory bridge between Mid-Northern Europe and the African continent. Furthermore, the islands of the Archipelago represent important nesting sites for several rare and threatened marine birds.

The flora and the vegetation of the Tuscan Archipelago reflect the Mediterranean context of the islands. Arrigoni et al. (2003) reviewed the main floristic traits of the Archipelago. According to this work, the Flora of the whole Tuscan Archipelago consists of circa 1300 taxa. Land surface area clearly influence floristic diversity of the islands, with species/area ratio expressed by a typically logarithmic curve,with a lower increase in floristic diversity as area increases in size (Arrigoni et al., 2003). Analyzing the chorology of the Tuscan Archipelago flora, there is a prevalence of Mediterranean and Tethydic elements, with a large presence of Euro-Mediterranean and Euro-Tethydic species, thus revealing a northern Mediterranean floristic combination (Euro-Mediterranean).The flora of the Archipelago appear highly influenced by the Tuscan flora, with a low penetration of Corso-Sardinian elements. However, analyzing the floristic endemism of these islands, a strong component of Corso-Sardinian endemism can be found across the western island, suggesting that the Archipelago represent a connection between Tuscan and Corso-Sardinian endemism. The Tuscan Archipelago account for 16 narrow endemics (Foggi et al. 2014), 17 endemics belonging to the Corso-Sardinian Dominion and 3 endemics belonging to the Tyrrhenian Dominion (Arrigoni et al. 2003). The endemic plant species can be found on Table 2.2, while the systematic position of *Saxifraga granulata* var. *brevicaulis* Sommier, cited in Arrigoni et al. (2003), but not treated in Foggi et al. (2014) need further investigations (Ferretti et al. 2014). Nowadays the floristic endemism of the Tuscan Archipelago appear directly and/or indirectly threatened by several factor generally highly related to human presence, such as tourism and agriculture and invasive species of plants and animals (Foggi et al. 2014). Between the threats indirectly associated with human activities the main consist in the loss of habitats surface due to abandonment of non-agricultural activities and change in vegetation dynamics. Human presence involves also impact related to density of infrastructures, tourism and recreational activities and gathering of rare species. The main threats due to alien species are those directly affecting habitats and/or species as competitors or predators. One of the major threats is represented by the over predation by feral or wild ungulates, such us feral goats (Foggi et al. 2014). Across the Invasive plants the main threats are represented by *Ailanthus altissimus*, *Carpobrothus sp. pl.*, *Acacia sp. pl.* and *Opuntia sp. pl.*

Tuscan Archipelago narrow endemics
Biscutella pichiana Raffaelli subsp. *Ilvensis* Raffaelli
Centaurea aethaliae (Sommier) Bég.
Centaurea dissecta Ten. var. *ilvensis* Sommier
Centaurea gymnocarpa Moris et De Not.
Crocus ilvensis Peruzzi & Carta
Festuca gamisansii Kerguélen subsp. *aethaliae*Signorini et Foggi
Limonium doriae (Somm.) Pign.
Limonium gorgonae Pign.
Limonium ilvae Pign.
Limonium planesiae Pign.
Limonium sommierianum (Fiori) Arrigoni
Linaria capraria Moris et De Not.
Mentha requienii Benth. subsp. *bistaminata* Mannocci et Falconcini
Romulea insularis Somm.
Silene capraria Sommier
Viola corsica Nyman subsp. *ilvensis* (W.Becker) Merxm.
Endemics belonging to the Corso-Sardinian Dominion
Arum pictum L. fil.
Borago pygmaea (DC.) Chater et W. Greuter
Carduus fasciculiflorus Viv.
Carex micro carpa Bertol. ex Moris
Festuca arundinacea Schreber subsp. *corsica* (Hack.) Kerguélen
Galium caprarium Natali
Hypericum hircinum L.
Limonium contortirameum (Mabille) Erben
Mentha suaveolens Ehrh. subsp. *insularis* (Req.) Greuter
Pancratium illyricum L.
Scrophularia trifoliata L.
Soleirolia soleirolii (Req.) Dandy
Stachys corsica Pers.
Stachys glutinosa L.
Trisetaria burnoufii (Req. exParl.) Banfi et Soldano
Urtica atrovirens Req. exLoisel.
Verbascum conocarpum Moris
Endemics belonging to the Tyrrhenian Dominion
Helichrysum litoreum Guss.
Ophrys exaltata Ten. subsp. *tyrrhena* (Gölz et Reinh.) Del Prete
Silene badaroi Breistr.

Table 2.2 Endemic species of the Tuscan Archipelago, divided between narrow endemics (16), endemics belonging to the Corso-Sardinian Dominion (17) and endemics belonging to the Tyrrhenian Dominion (3).

The vegetation of the Tuscan Archipelago reflects the high level of human influence that characterized these ecosystems in the last centuries. Human influence caused the almost total vanishing of the original forest vegetation and the development of cultivated and urbanized areas on the islands (Arrigoni et al. 2003). Nowadays true woodlands can be only found on the islands of Elba, Giannutri and, really poorly represented, on Gorgona. Elba still host quite good extension of holm oak (*Quercus ilex)* forests and old chestnut (*Castanea sativa*) plantation, the latter on the slopes of the Monte Capanne (Foggi et al. 2006). Few extension of holm oak forest can also be found on Giglio and Gorgona. However, the dominant vegetation formation in the whole Archipelago is represented by the typical Mediterranean *macchia*, in all his stages of degradation. Thus going from the closest formation dominated by ericaceae such us *Erica sp. pl.* and *Arbutus unedo*, to low macchia and garrigue formations. Finally also perennial and temporal grassland are well represented, even thou the latter are mainly found scattered in mosaic formations with open macchia formations.

The complex mosaic of vegetation types that can be found on the islands determine a great habitat diversity. This lead to presence in the Tuscan Archipelago of a great number of habitat of conservationist value, as well described by the great occurrence of Natura2000 habitat. Only in the Island of Elba 27 different Natura2000 habitats have been listed (Viciani et al., in press). Four of these are habitats of priority interest: the coastal dunes with *Juniperus spp.* (cod. 2250*); the Mediterranean temporary ponds (cod. 3170*); the pseudo-steppe with grasses and annuals of the *Thero-Brachypodietea* (cod. 6220*). Many of these habitats are represented also in the smallest islands, which often host a quite heterogeneous habitat mosaic. The alluvial forests with *Alnus glutinosa* and *Fraxinus excelsior* (*Alno-Padion*, *Alnion incanae*, *Salicion albae*) (cod. 91E0*) are instead found only in Elba.

As common in Mediterranean basin landscapes and ecosystems, human influence represented, and still does, an essential factor in the biological evolution of the islands of the Tuscan Archipelago. The human influence on the natural environment has started around 6,000 years b.p. and has become massive since the Roman times (2,400 years b.p.). The islands, especially Elba, were largely exploited for mineral extraction, particularly iron, since times of ancient Greek and Roman colonization. These activities on the islands also lead to an ancient overexploitation of the woodlands of the islands, determining the disappearance of original forest vegetation. During recent history, the human activities on the islands were mainly related, as well as mining, to a subsistence economic model, mainly driven by agriculture and livestock farming, principally of goats and sheep (Repetti 1833-1840).

However, in the last century, like many other Mediterranean islands (Delanoe et al. 1996), the Tuscan Archipelago islands have undergone a deep change of their landscape and land-uses, from agro-forestry to tourism activities. This led firstly to a decline in resident population, and in the abandonment of traditional agricultural practices and land uses, and secondarily to a shift in the land use, with a massive emergence of touristic recreational facilities and in the increase of human seasonal fluxes.

These socio-economic changes play a key role in the themes addressed within this work. Indeed changes in the disturbance regimes, in the land-use, and climate are driving factors in determining plant invasions (Lonsdale 1999; Mooney and Hobbs 2000). Moreover these factors are expected to be much powerful in the islands. In spite it represents a valuable economic source for local population, mass tourism raise many sustainability issues (Brundu, 2013). Particularly to biological invasion this shift has changed human impact on the insular biota: plants are not introduced for alimentary reasons such as in the Neolithic (about 5.500 b.p.) but as ornaments (Hulme 2004). It is noteworthy that almost half of all plant introductions to Mediterranean islands are related to gardens and landscaping associated with tourist developments and housing and gardens estate. It therefore follows that this is likely to be a major source of naturalised species (Brundu 2013 and references therein).

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3. DEVELOPING BASIC KNOWLEDGE

The importance of national and regional checklists for understanding invasion was demonstrated by a wide series of studies carried out thanks to the data provided by such catalogues (Pysek et al. 2012). Currently updated Checklists of alien plants of specific areas are considered valuable tools to provide standardized information for such areas, and to evaluate the changes that occur in the invasive status over time, allowing the invasion to be monitored at a local scale (Pyšek et al. 2012). For example European catalogues provided essential data used to analyze invasion patterns at the continental level, including cross-taxonomic evaluation of determinants of regional levels of invasion, distribution of alien species in habitats, assessment of ecological and economic impacts of alien species in Europe and risk-assessment for plants based on habitat mapping (Pyšek et al. 2012 and references therein).

The importance of basic knowledge on biological invasion it is also underlined in the CBD strategic goals for 2020 on biological invasion. These state that by 2020, invasive alien species and pathways should be identified and prioritized, priority species controlled or eradicated, and measures should be put in place to manage pathways to prevent their introduction and establishment. It is striking that the production of updated checklists of alien species introduced on a territory and the identification of pathways and distributions underlie the adoption and prioritization of efficient management actions.

The phenomenon of biological invasions and of introductions of alien species in the Tuscan Archipelago is not new in the floristic researches on the Archipelago. Occasionally, the presence of alien plants was recorded since the earlier contribution on the flora of the Tuscan archipelago (starting from around mid-1850s) towards the most recent, as the floristic lists of Montecristo (Paoli and Romagnoli 1976) and Elba Island (Fossi Innamorati 1983, 1989, 1991, 1994, 1997), the contributions by Baldini (1998, 2000, 2001) for Giglio, Pianosa and Giannutri and by Foggi et al. (2001) for Capraia and the islets' floras (Baldini 1990, 1991; Foggi et al. 2009). Furthermore, especially in the last few years several single contributions with records of single or few species were produced for the Tuscan Archipelago.

Since 1990s the Tuscan Archipelago National Park started several project aimed to the management of some well-known invasive species. They were mostly developed within the EU LIFE program. The project LIFE - Natura B4-3200/97/271 "Capraia and the smaller islands of Tuscany: conservation of biodiversity" focused on the eradication of black rat (*Rattus rattus*) on several islets of the Archipelago and of alien plants, *Ailanthus altissimus* and *Carpobrotus sp. pl*. from Capraia. Later project LIFE04NAT/IT/000172 "Tuscan Islands: new actions for sea birds and habitats” again was focused on the eradication of black rat from the island of Giannutri and the islet of La Scola, as well as the control of the feral cat (*Felix catus*) population and other problematic invasive, but native, plant species (i.e *Pinus halepensis*), on Pianosa. The eradication of black rat and *Ailanthus altssima on Montecristo* was also the aim of LIFE08 NAT/IT/000353 “Montecristo island 2010: eradication of invasive alien flora and fauna and protection of species and habitat in the Tuscan archipelago”. This project was focused also on the eradication of invasive alien species on Pianosa (i.e *Ailanthus altissimus*; *Carpobrotus sp. pl.; Acacia sp. pl.* and *Senecio angulatus*). Furthermore, a LIFE project (LIFE13-NAT_IT_000471) has recently started focusing habitat restorations and on the eradication of several plant species in Montecristo and Giannutri, as well as on eradication of black rat on Pianosa.

In spite of this attention to the matter since the last years it was completely lacking a comprehensive approach to the problem, particularly to alien plants. Indeed the records of alien plant available in the literature regarded mainly few well known species, occasionally found during the field explorations aimed to the depiction of native flora. Furthermore it was completely lacking any distributive or ecological investigation on the phenomenon, resulting in a massive lack of such kind of information. Also the actions undertaken by TANP were mainly focused on an essential, but still restricted, pool of invasive plants.

Arrigoni and Viegi (2011) published a booklets on the alien plants of Tuscany, unfortunately this paper reported old data, without a precise geographic information and with a classification of the “invasive status” not properly in accordance with the standard nomenclature.

Finally starting from 2011, within the CoREM framework (Cooperazione delle Reti Ecologiche nel Mediterraneo - Cooperation of Ecological Networks in the Mediterranean), TANP began a significant comprehensive project aimed to an in-deep investigation of the issues of biological invasion. The project was dedicated to the production of checklists of the alien biota introduced in the TANP territories and in the definition of black list of particularly invasive species.

Given the high importance of basic knowledge and the implications that it can have on efficient control of the phenomenon, with this PhD thesis, and within TANP CoREM project, we started a research project devoted to updating the current knowledge of the alien flora of the Tuscan Archipelago. The project led to the publication of some minor contributions and finally to the production of the "check list of alien flora of the Tuscan Archipelago".

The work aimed not only to record the presence of alien species across the island of Tuscan Archipelago, but an essential aim was to assess the invasive behavior of these species. Toward these aim we decide to assess the "*invasive status*" of the species according to the terminology introduced by Richardson et al (2000) and Pysek et al. (2004). The concepts and terms definition introduced by the authors offer the possibility to characterize, with fair precision, species' residence and invasion status, aiming to allow a better understanding between taxonomists and ecologists and more detailed comparative analyses of alien floras of various regions of the world (Pysek et al. 2004).

According to this terminology, we distinguished the species between:

> *Casual alien plants* - Alien plants that may flourish and even reproduce occasionally outside cultivation in an area, but that eventually die out because they do not form self-replacing populations, and rely on repeated introductions for their persistence.
>
> *Naturalized plants* - Alien plants that sustain self-replacing populations for at least 10 years without direct intervention by people (or in spite of human intervention) by recruitment from seed or ramets (tillers, tubers, bulbs, fragments, etc.) capable of independent growth.
>
> *Invasive plants* - Invasive plants are a subset of naturalized plants (Fig. 2) that produce reproductive offspring, often in very large numbers, at considerable distances from the parent plants, and thus have the potential to spread over a large area.

The collection of the data required to the development of the checklist embraced two main phases. The first phases regarded the gathering of all literature available on alien plant in Tuscan Archipelago, thus including all the works cited above on the floras of the Tuscan Archipelago up to the several single records produced more recently. All this information was stored in a DataBase, which allowed to query the information collected, highlighting eventual lacking of information and allowing to concentrate the investigations on the less known taxa or areas of TANP. Subsequently to the gathering of all the literature, started a massive field exploratory phase. This work aimed both to improve the collection of presence data, especially for islands or species less known, and to evaluate the “invasion status” of the species across the islands. Thus, we visited all the islands of the Archipelago several time, in different periods of the year, aiming to cover the time variability in the phenology of the species. This collection of data was an essential part of the activities undertaken in the years 2012-2013, and led to the production of several minor contributions mainly reporting new records of alien species (see sections 3.1-2).

Throughout 2013 and early 2014 we started an essential phase of synthesis of the data collected. This work passed through the cross-checking of a huge number of herbarium specimens of alien species stored in the herbarium of Florence. On the one hand, this work aimed to confirm the suitability of oldest records, but it also allowed us to enhance our general knowledge on these species and to evaluate their presence in the Tuscan Archipelago across time.

We encountered and faced several issues, mainly regarding the unclear taxonomic position of certain species, misused names and wrong identification found across literature. The two problems were actually often linked, because difficulties in correct identifications in past often led to use of wrong or invalid names or to the misinterpretation of the species identity. These issues regarded mainly the groups of Australian *Acacias*, *Amaranthus sp pl.* and *Erigeron sp. pl.*

Finally one last phase of the production of the checklist, regarded the collection of general information on the species, such as residence time, biological form, the introduction pathway, and the study of variation of these traits among time, aiming to identify important trends in the evolution of plant invasions in the Tuscan Archipelago.

The results of this researches, consisting in the updated checklist of alien plant of the Tuscan Archipelago and in the evaluation of its main traits were finally published in 2014 (Section 3.3).

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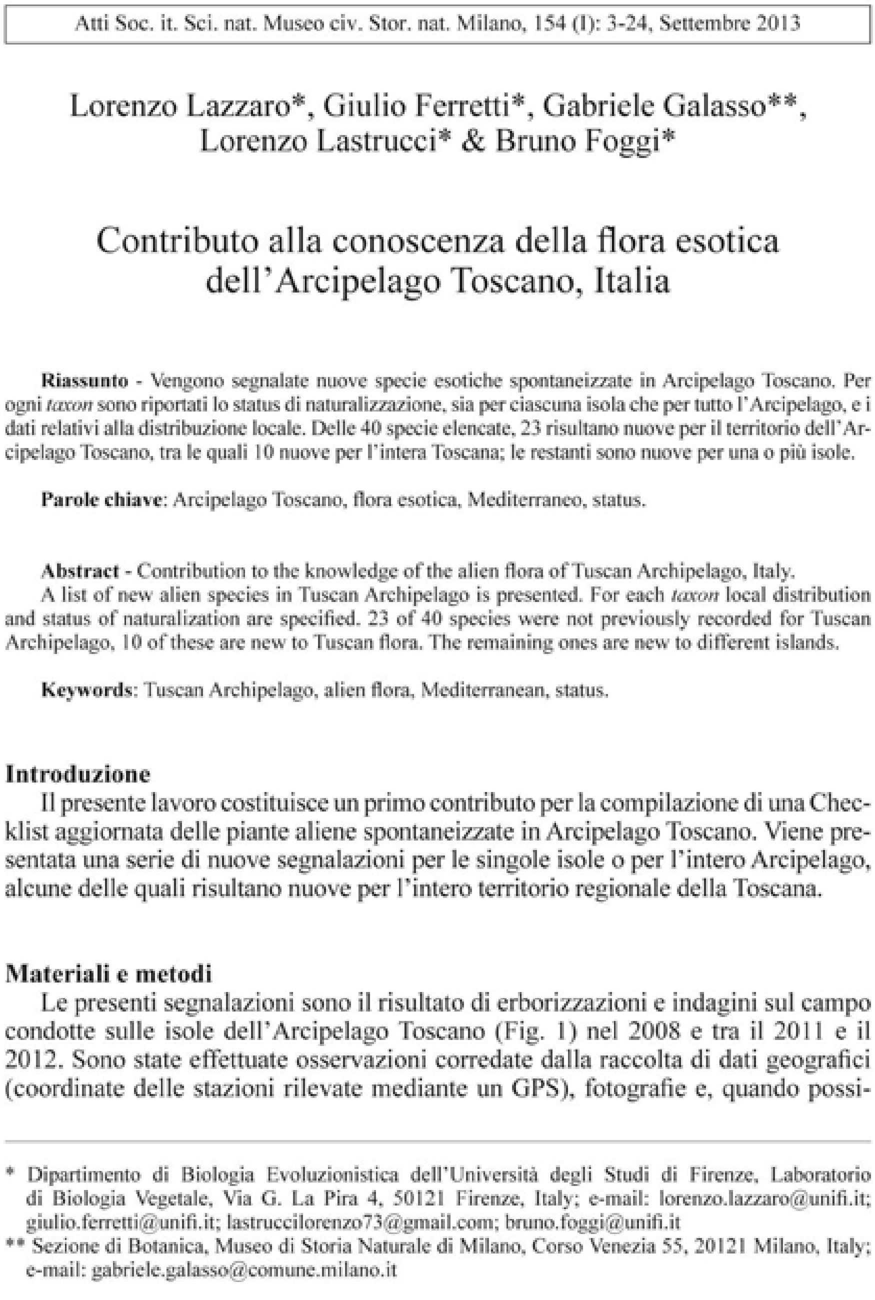
3.1 Contribution to the knowledge of the alien flora of Tuscan Archipelago, Italy.



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Contributo alla conoscenza della flora esotica dell'Arcipelago Toscano, Italia

Riassunto - Vengono segnalate nuove specie esotiche spontaneizzate in Arcipelago Toscano. Per ogni *taxon* sono riportati lo status di naturalizzazione, sia per ciascuna isola che per tutto l'Arcipelago, e i dati relativi alla distribuzione locale. Delle 40 specie elencate, 23 risultano nuove per il territorio dell'Arcipelago Toscano, tra le quali 10 nuove per l'intera Toscana; le restanti sono nuove per una o più isole.

Parole chiave: Arcipelago Toscano, flora esotica, Mediterraneo, status.

Abstract - Contribution to the knowledge of the alien flora of Tuscan Archipelago, Italy.

A list of new alien species in Tuscan Archipelago is presented. For each *taxon* local distribution and status of naturalization are specified. 23 of 40 species were not previously recorded for Tuscan Archipelago, 10 of these are new to Tuscan flora. The remaining ones are new to different islands.

Keywords: Tuscan Archipelago, alien flora, Mediterranean, status.

Introduzione

Il presente lavoro costituisce un primo contributo per la compilazione di una Checklist aggiornata delle piante aliene spontaneizzate in Arcipelago Toscano. Viene presentata una serie di nuove segnalazioni per le singole isole o per l'intero Arcipelago, alcune delle quali risultano nuove per l'intero territorio regionale della Toscana.

Materiali e metodi

Le presenti segnalazioni sono il risultato di erborizzazioni e indagini sul campo condotte sulle isole dell'Arcipelago Toscano (Fig. 1) nel 2008 e tra il 2011 e il 2012. Sono state effettuate osservazioni corredate dalla raccolta di dati geografici (coordinate delle stazioni rilevate mediante un GPS), fotografie e, quando possi-

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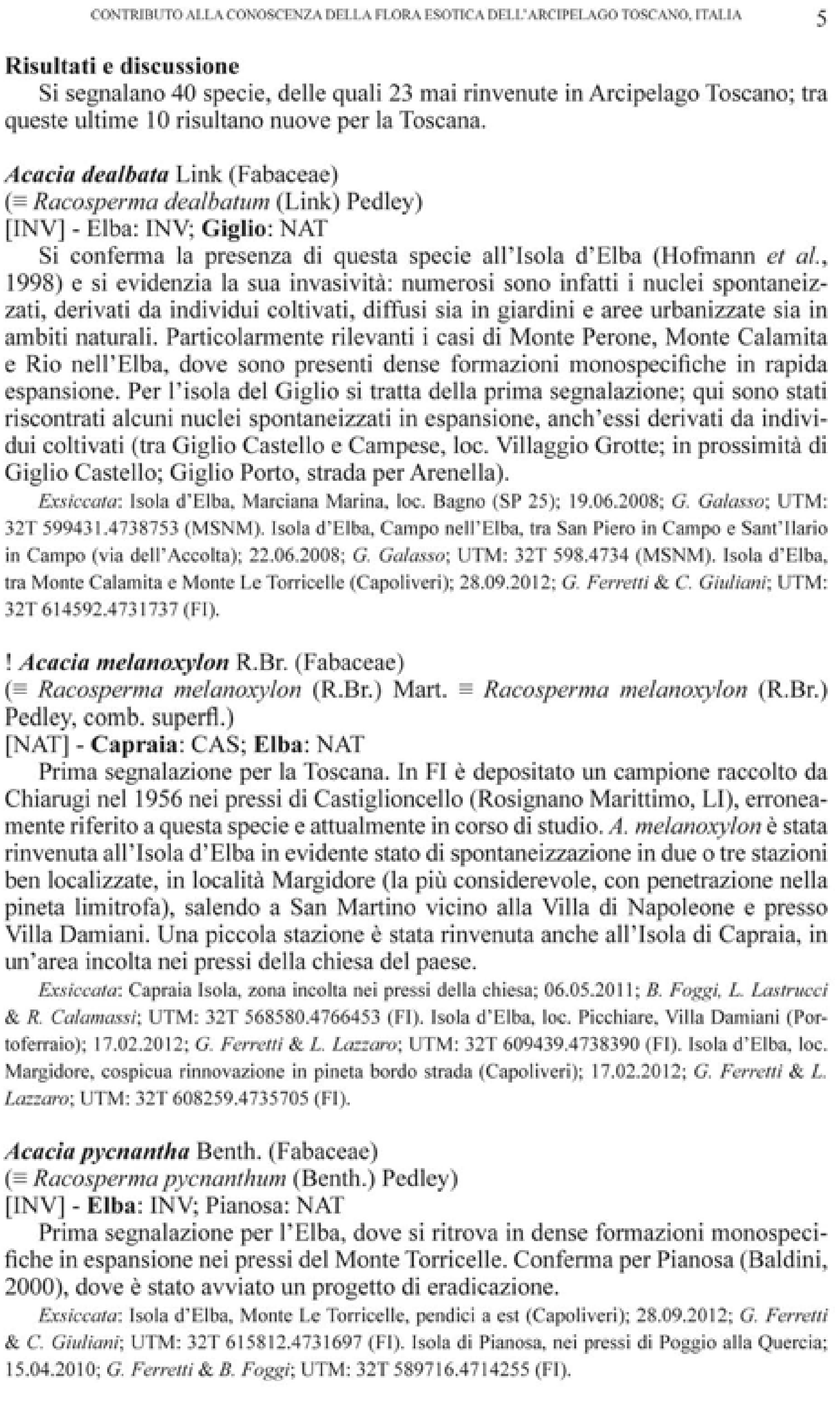
bile, campioni di erbario. Questi ultimi sono conservati presso l'Erbario Centrale di Firenze (FI) e presso l'Erbario del Museo di Storia Naturale di Milano (MSNM).

La nomenclatura utilizzata segue Banfi *et al.* (2009) e Celesti-Grapow *et al.* (2010), tranne alcuni casi opportunamente specificati. Le specie sono riportate in ordine alfabetico e, laddove necessario, sono indicati i principali sinonimi. Per ogni entità sono fornite le informazioni su presenza, distribuzione e status di naturalizzazione nelle singole isole; per la valutazione di quest'ultimo sono state adottate le definizioni di Richardson *et al.* (2000) e Pyšek *et al.* (2004): casuale, naturalizzato, invasivo. In parentesi quadra è indicato lo status in Arcipelago (considerando il valore più alto tra quelli delle singole isole, secondo l'ordine invasivo > naturalizzato > casuale); segue l'elenco alfabetico delle isole sulle quali la specie è stata rinvenuta allo stato spontaneo, ciascuna con il grado di naturalizzazione presentato su di essa dalla specie in questione. I dati sulla presenza in Toscana sono stati desunti da Celesti-Grapow *et al.* (2009, 2010) e Arrigoni & Viegi (2011). Viene utilizzata la seguente simbologia:
! = prima segnalazione per la Toscana,
* = prima segnalazione per l'Arcipelago Toscano,
isola in grassetto = prima segnalazione per l'isola,
CAS = casuale,
NAT = naturalizzata,
INV = invasiva,
NR = specie non ritrovata, presumibilmente estinta localmente.

Laddove siano stati raccolti campioni si cita la dicitura del cartellino d'erbario, nel seguente ordine: località; data di raccolta; raccoglitore/i; coordinate cartografiche nel sistema di riferimento UTM (WGS84) ed erbario in cui è stato depositato il saggio.

Fig. 1 - Inquadramento geografico dell'Arcipelago Toscano. / Geographical overview of Tuscan Archipelago.



Risultati e discussione

Si segnalano 40 specie, delle quali 23 mai rinvenute in Arcipelago Toscano; tra queste ultime 10 risultano nuove per la Toscana.

Acacia dealbata Link (Fabaceae)
(≡ *Racosperma dealbatum* (Link) Pedley)
[INV] - Elba: INV; **Giglio**: NAT

Si conferma la presenza di questa specie all'Isola d'Elba (Hofmann *et al.*, 1998) e si evidenzia la sua invasività: numerosi sono infatti i nuclei spontaneizzati, derivati da individui coltivati, diffusi sia in giardini e aree urbanizzate sia in ambiti naturali. Particolarmente rilevanti i casi di Monte Perone, Monte Calamita e Rio nell'Elba, dove sono presenti dense formazioni monospecifiche in rapida espansione. Per l'isola del Giglio si tratta della prima segnalazione; qui sono stati riscontrati alcuni nuclei spontaneizzati in espansione, anch'essi derivati da individui coltivati (tra Giglio Castello e Campese, loc. Villaggio Grotte; in prossimità di Giglio Castello; Giglio Porto, strada per Arenella).

Exsiccata: Isola d'Elba, Marciana Marina, loc. Bagno (SP 25); 19.06.2008; *G. Galasso*; UTM: 32T 599431.4738753 (MSNM). Isola d'Elba, Campo nell'Elba, tra San Piero in Campo e Sant'Ilario in Campo (via dell'Accolta); 22.06.2008; *G. Galasso*; UTM: 32T 598.4734 (MSNM). Isola d'Elba, tra Monte Calamita e Monte Le Torricelle (Capoliveri); 28.09.2012; *G. Ferretti* & *C. Giuliani*; UTM: 32T 614592.4731737 (FI).

! ***Acacia melanoxylon*** R.Br. (Fabaceae)
(≡ *Racosperma melanoxylon* (R.Br.) Mart. ≡ *Racosperma melanoxylon* (R.Br.) Pedley, comb. superfl.)
[NAT] - **Capraia**: CAS; **Elba**: NAT

Prima segnalazione per la Toscana. In FI è depositato un campione raccolto da Chiarugi nel 1956 nei pressi di Castiglioncello (Rosignano Marittimo, LI), erroneamente riferito a questa specie e attualmente in corso di studio. *A. melanoxylon* è stata rinvenuta all'Isola d'Elba in evidente stato di spontaneizzazione in due o tre stazioni ben localizzate, in località Margidore (la più considerevole, con penetrazione nella pineta limitrofa), salendo a San Martino vicino alla Villa di Napoleone e presso Villa Damiani. Una piccola stazione è stata rinvenuta anche all'Isola di Capraia, in un'area incolta nei pressi della chiesa del paese.

Exsiccata: Capraia Isola, zona incolta nei pressi della chiesa; 06.05.2011; *B. Foggi, L. Lastrucci* & *R. Calamassi*; UTM: 32T 568580.4766453 (FI). Isola d'Elba, loc. Picchiare, Villa Damiani (Portoferraio); 17.02.2012; *G. Ferretti* & *L. Lazzaro*; UTM: 32T 609439.4738390 (FI). Isola d'Elba, loc. Margidore, cospicua rinnovazione in pineta bordo strada (Capoliveri); 17.02.2012; *G. Ferretti* & *L. Lazzaro*; UTM: 32T 608259.4735705 (FI).

Acacia pycnantha Benth. (Fabaceae)
(≡ *Racosperma pycnanthum* (Benth.) Pedley)
[INV] - **Elba**: INV; Pianosa: NAT

Prima segnalazione per l'Elba, dove si ritrova in dense formazioni monospecifiche in espansione nei pressi del Monte Torricelle. Conferma per Pianosa (Baldini, 2000), dove è stato avviato un progetto di eradicazione.

Exsiccata: Isola d'Elba, Monte Le Torricelle, pendici a est (Capoliveri); 28.09.2012; *G. Ferretti* & *C. Giuliani*; UTM: 32T 615812.4731697 (FI). Isola di Pianosa, nei pressi di Poggio alla Quercia; 15.04.2010; *G. Ferretti* & *B. Foggi*; UTM: 32T 589716.4714255 (FI).

* ***Acacia saligna*** (Labill.) H.L.Wendl. (Fabaceae)
(≡ *Mimosa saligna* Labill. ≡ *Racosperma salignum* (Labill.) Pedley)
[NAT] - **Pianosa**: NAT

Prima segnalazione per l'Arcipelago Toscano, ove abbiamo osservato alcuni individui naturalizzati sull'Isola di Pianosa. In Toscana è indicata come naturalizzata da Celesti-Grapow *et al.* (2009, 2010). Analogamente ad *A. pycnantha*, è stato avviato un intervento di eradicazione.

Exsiccata: Isola di Pianosa, tra la Torretta S. Marco e Punta Libeccio; 03.03.1999; *R.M. Baldini* & *L. Vivona* (FI! sub *Acacia pycnantha*). Isola di Pianosa, nei pressi di Poggio alla Quercia; 15.04.2010; *G. Ferretti* & *B. Foggi*; UTM: 32T 589721.4714258 (FI). Isola di Pianosa, nei pressi di Punta del Segnale; 15.04.2010; *G. Ferretti* & *B. Foggi*; UTM: 32T 587683.4713853 (FI).

Aeonium arboreum (L.) Webb & Berthel. (Crassulaceae)
(≡ *Sempervivum arboreum* L.)
[NAT] - **Capraia**: NAT; **Elba**: CAS; **Giannutri**: CAS; Giglio: NAT; Gorgona: CAS; Pianosa: NAT

Prima segnalazione per l'Isola d'Elba, Capraia e Giannutri; conferma per Giglio (Baldini, 1998), Gorgona (Arrigoni & Viegi, 2011) e Pianosa (Baldini, 2000). Questa specie presenta la maggiore diffusione a Pianosa, dove si rinviene piuttosto comunemente nelle zone antropizzate, su buona parte dei muri a secco che recingono alcune aree del paese e in corrispondenza della Torre di Babele. Nel resto dell'Arcipelago manifesta minore vigoria, pur essendo abbastanza diffusa a Capraia e al Giglio; su quest'ultima isola mostra evidenti segni di rinnovazione, riproducendosi per seme nelle tasche terrose della scogliera tra Giglio Porto e Le Cannelle.

Anredera cordifolia (Ten.) Steenis (Basellaceae)
(≡ *Boussingaultia cordifolia* Ten.)
[NAT] - Capraia: NAT; **Elba**: NAT; Giglio: NAT; Gorgona: NAT; **Pianosa**: NAT

Specie già segnalata all'Isola del Giglio (Baldini, 1998), Gorgona (Rizzotto, 2011) e Capraia (Arrigoni & Viegi, 2011). Si aggiungono qui le prime segnalazioni per Elba (Pomonte, Capoliveri) e Pianosa (Porcile, Sembolello, il Giudice). La specie è solitamente legata agli ambienti antropici, dove si espande con notevole vigoria addossandosi alle infrastrutture e alla vegetazione naturale.

Exsiccata: Isola d'Elba, Pomonte, parte bassa del paese, su rudere abbandonato; 28.09.2012; *G. Ferretti* & *C. Giuliani*; UTM: 32T 591877.4733458 (FI). Isola di Pianosa, nei pressi del Giudice; 28.09.2012; *L. Lazzaro* & *B. Foggi*; UTM: 32T 589240.4714489 (FI).

Asclepias fruticosa L. (Apocynaceae)
(≡ *Gomphocarpus fruticosus* (L.) W.T.Aiton)
[NAT] - **Elba**: CAS; Giglio: NAT

Prima segnalazione per l'Isola d'Elba, dove sinora è stata ritrovata un'unica stazione con diversi individui spontaneizzati nei pressi di Porto Azzurro, sulla strada per il Monserrato. La pianta è stata segnalata all'Isola del Giglio da Baldini (1998).

Exsiccata: Isola d'Elba, Porto Azzurro, strada per il Monserrato; 18.02.2012; *G. Ferretti* & *L. Lazzaro*; UTM: 32T 614600.4737060 (FI).

Austrocylindropuntia subulata (Muehlenpf.) Backeb. (Crassulaceae) (Fig. 2)
(≡ *Pereskia subulata* Muehlenpf. ≡ *Opuntia subulata* (Muehlenpf.) Engelm. ≡ *Cylindropuntia subulata* (Muehlenpf.) F.M.Knuth)

Fig. 2 - Isola di Pianosa: *Austrocylindropuntia subulata* in prossimità dell'abitato. / Pianosa Island: *Austrocylindropuntia subulata* near the village (Foto / Photo B. Foggi).

[NAT] - **Capraia**: NAT; Elba: NAT; **Giannutri**: CAS; Giglio: CAS; **Montecristo**: CAS; **Pianosa**: NAT

Prima segnalazione per Capraia (al Faro), Giannutri (area residenziale), Montecristo (nei pressi della Villa Reale) e Pianosa (in Paese, zona Teglia); conferma dei dati di Guiggi (2008) per Elba e Giglio. Si tratta di una pianta con elevata capacità propagativa, per cui in prossimità degli individui coltivati si registra quasi sempre la sua spontaneizzazione (Arrigoni & Viegi, 2011).

! ***Callitropsis glabra*** (Sudw.) D.P.Little (Cupressaceae)
(≡ *Cupressus glabra* Sudw. ≡ *Cupressus arizonica* Greene subsp. *glabra* (Sudw.) A.E.Murray ≡ *Hesperocyparis glabra* (Sudw.) Bartel ≡ *Neocupressus arizonica* (Greene) de Laub. var. *glabra* (Sudw.) de Laub. = *Cupressus arizonica* auct., non Greene)
[NAT] - **Elba**: NAT

Prima segnalazione per la Toscana. Specie ampiamente utilizzata a scopo ornamentale e per alberature, finora mai trovata allo stato spontaneo anche a causa della confusione con la congenere *C. arizonica*, rispetto alla quale merita autonomia a rango specifico (Little, 2006). Si ricorda che i cipressi del nuovo mondo sono stati separati su base filogenetica da quelli del vecchio mondo. Rispetto alle trattazioni di Adams *et al.* (2009), de Laubenfels (2009) e de Laubenfels *et al.* (2012), che separano ulteriormente i cipressi del nuovo mondo (*Hesperocyparis* Bartel & R.A.Price ≡ *Neocupressus* de Laub., nom. illeg.) da *Callitropsis nootkatensis* (D.Don) Oerst. (sempre nordamericana), noi preferiamo seguire Little (2006), che li mantiene uniti sotto il genere *Callitropsis* Oerst., poiché esistono ben tre ibridi a vario grado di fertilità tra i due presunti generi; a parte rimane *Xanthocyparis vietnamensis* Farjon & T.H.Nguyên del sudest asiatico.

Vengono qui segnalati alcuni nuclei in evidente stato di naturalizzazione all'Isola d'Elba, tra il Monte Le Torricelle e il Monte Calamita, dove probabilmente *C. glabra* è stata introdotta coi rimboschimenti degli anni '50-'70 del secolo scorso (Gatteschi & Arretini, 1989, sub *Cupressus arizonica*). La specie è presente come *culta* anche in altre isole, ma non abbiamo riscontrato altri casi di spontaneizzazione.

Exsiccata: Isola d'Elba, tra M. Le Torricelle e M. Calamita (Capoliveri); 17.02.2012; *G. Ferretti & L. Lazzaro*; UTM: 32T 615106.4731677 (FI).

* ***Campsis radicans*** (L.) Bureau (Bignoniaceae)
(≡ *Bignonia radicans* L. ≡ *Campsis radicans* (L.) Seem., comb. superfl. ≡ *Tecoma radicans* (L.) Juss.)
[NAT] - **Giglio**: NAT

Ampiamente utilizzata come rampicante ornamentale, è presumibilmente coltivata da molto tempo in quasi tutte le isole dell'Arcipelago, ma non è mai stata rinvenuta spontaneizzata. Si segnalano qui alcuni nuclei naturalizzati sull'Isola del Giglio, presso Giglio Campese e Giglio Porto; uno, in particolare, è stato rinvenuto poco sopra Giglio Porto, al margine della strada per Castello, a notevole distanza dai centri abitati e quindi completamente affrancato dagli individui coltivati. In Toscana è indicata da Arrigoni & Viegi (2011) come occasionale nella Macchia lucchese.

Exsiccata: Isola del Giglio, poco sopra Giglio Porto, al margine della strada per Giglio Castello; 15.07.2011; *G. Ferretti & B. Foggi*; UTM: 32 T 656836.4692398 (FI).

Carpobrotus acinaciformis (L.) L.Bolus (Aizoaceae) (Figg. 3-4)
(≡ *Mesembryanthemum acinaciforme* L.)
[INV] - Capraia: INV; Elba: INV; Giannutri: INV; Giglio: INV; **Gorgona**: INV; Montecristo: CAS; Pianosa: INV

Prima segnalazione per l'Isola di Gorgona, in località Torre Nuova, conferma nelle altre isole (Baldini 1998, 2000, 2001; Foggi *et al.*, 2001; Fossi Innamorati, 1983). La stazione di Villa Reale di Montecristo, citata da Paoli & Romagnoli (1976) e non più rinvenuta negli ultimi anni, è stata confermata dalla nostra indagine che ha permesso di individuare alcune plantule di un anno sui muri a secco di Cala Maestra. Il recente intervento di derattizzazione dell'isola e il controllo della pressione del carico pascolivo da parte delle capre potrebbe averne favorito la rinnovazione da seme, finora ostacolata da tali presenze.

Carpobrotus edulis (L.) N.E.Br. (Aizoaceae) (Figg. 3-5)
(≡ *Mesembryanthemum edule* L.)
[INV] - Elba: NAT; **Giannutri**: INV; **Giglio**: INV

Specie spesso confusa o inclusa in *C. acinaciformis* e quindi probabilmente sottostimata nei precedenti lavori. In accordo con Hartmann (2001) manteniamo separate le due entità e segnaliamo la presenza di alcune popolazioni di *C. edulis* a Giannutri (in prossimità del vecchio aeroporto; al Faro, presso lo scoglio dei Grottoni) e al Giglio (Giglio Porto, scogliere a nord; spiaggia di Cannelle; Campese, scogliere nei pressi della Torre). Citata genericamente per l'Elba da Arrigoni & Viegi (2011).

Fig. 3 - Isola di Gorgona: fiori di *Carpobrotus acinaciformis* e foglie di *C. edulis* nelle vicinanze del porto. / Gorgona Island: flowers of *Carpobrotus acinaciformis* and leaves of *C. edulis* near the heaven. (Foto / Photo G. Ferretti).

Fig. 4 - Isola di Giannutri: *Carpobrotus acinaciformis* e *C. edulis* presso Punta San Francesco. / Giannutri Island: *Carpobrotus acinaciformis* and *C. edulis* at Punta San Francesco. (Foto / Photo G. Ferretti).

Fig. 5 - Isola di Giannutri: *Carpobrotus edulis* sulle scogliere presso Cala Spalmatoio. / Giannutri Island: *Carpobrotus edulis* on the rocky shore at Cala Spalmatoio. (Foto / Photo G. Ferretti).

* ***Cortaderia selloana*** (Schult. & Schult.f.) Asch. & Graebn. (Poaceae)
(≡ *Arundo selloana* Schult. & Schult.f.)
[INV] - **Elba**: INV

Prima segnalazione per l'Arcipelago. Si tratta di una specie ampiamente utilizzata a scopo ornamentale che ha dimostrato un forte carattere invasivo in altre aree mediterranee (Domènech & Vilà, 2007). Allo stato spontaneo è diffusa in numerose località dell'Elba; di particolare rilievo un nucleo molto esteso nei pressi di Portoferraio, dove diverse decine di individui hanno colonizzato un'area incolta soggetta a ristagno idrico. Citata per la Toscana continentale da Arrigoni & Viegi (2011).

Exsiccata: Isola d'Elba, Portoferraio, in periferia, lungo la SP 24, area incolta soggetta a ristagno idrico; 28.09.2012; *G. Ferretti* & *C. Giuliani*; UTM: 32T 606866.4739561 (FI).

! ***Cotyledon orbiculata*** L. (Crassulaceae)
[CAS] - **Capraia**: CAS; **Elba**: CAS; **Giannutri**: CAS; **Giglio**: CAS; **Pianosa**: CAS

Questa specie, mai segnalata allo stato spontaneo in Toscana, è ampiamente utilizzata a scopo ornamentale in Arcipelago Toscano, dove risulta spontaneizzata sulle seguenti isole: Capraia, pendice lungo la strada tra il porto e il paese; Elba, S. Andrea, sulle scogliere; Giannutri, paese, sopra al campo sportivo; Giglio, lungo la strada per l'Arenella; Pianosa, La Centrale. Al momento le giovani piante si mantengono nelle vicinanze dei nuclei di origine e non si sono mai osservate significative presenze in contesti naturali. Tuttavia, trattandosi di specie con elevata capacità propagativa, la situazione è da tenere sotto osservazione.

Exsiccata: Isola di Pianosa, in paese; 03.07.2011; *B. Foggi*; UTM: 32T 590133.4715530 (FI).

! ***Crassula lycopodioides*** Lam. (Crassulaceae)
[CAS] - **Elba**: CAS; **Giannutri**: CAS; **Giglio**: CAS

Prima segnalazione per la Toscana. Osservata per la prima volta spontaneizzata in Arcipelago Toscano nel 2012, sempre in situazioni antropizzate, ma con massiccia produzione di propaguli affrancati dagli individui coltivati. La si riporta per l'Isola d'Elba (Pomonte, nei pressi dell'Hotel Corallo), Giannutri (in paese) e Giglio (Giglio Porto, presso la fontana di Via del Castello).

Exsiccata: Isola d'Elba, Pomonte, nei pressi dell'Hotel Corallo (Marciana); 30.05.2012; *G. Ferretti* & *L. Lazzaro*; UTM: 32T 658051.4691494 (FI). Isola di Giannutri, in paese, nei pressi dei generatori; 22.06.2012; *G. Ferretti* & *L. Lazzaro*; UTM: 32T 673249.4680131 (FI).

Cupressus sempervirens L. (Cupressaceae)
[NAT] - Capraia: CAS; Elba: NAT; Giglio: NAT; Gorgona: CAS; **Pianosa**: NAT

Prima segnalazione per l'Isola di Pianosa, dove è presente una popolazione in evidente stato di naturalizzazione lungo la strada a nordest del Cardon, verso il cimitero dei carcerati. Si conferma nelle altre isole: coltivata a Giannutri e Montecristo, spontaneizzata altrove (Baldini, 1998, 2001; Foggi *et al.*, 2001; Fossi Innamorati, 1983; Paoli & Romagnoli, 1976; Rizzotto, 2011; Arrigoni & Viegi, 2011).

Exsiccata: Isola di Pianosa, strada a nord del Cardon; 28.09.2012; *B. Foggi* & *L. Lazzaro*; UTM: 32T 587608.4715095 (FI).

! ***Datura wrigthii*** Regel (Solanaceae)
(= *Datura inoxia* auct. p.p., non Mill.)
[CAS] - **Giannutri**: CAS

Prima segnalazione per la Toscana, anche se probabilmente vanno riferite a questa specie quasi tutte (se non tutte) le precedenti segnalazioni di *D. inoxia* (Lambinon, 2006; Verloove, 2008; Verloove *et al.*, 2010; Banfi & Galasso, 2010). In FI, ad esempio, è stato così rideterminato un campione proveniente dall'Argentario. In Arcipelago Toscano la segnaliamo spontaneizzata a Giannutri (Cala Spalmatoio e presso i generatori elettrici). Infine riteniamo importante segnalare la sua coltivazione al Giglio (Giglio Campese), in quanto specie potenzialmente invasiva e, analogamente alle altre entità del genere *Datura*, molto pericolosa per la salute se utilizzata in modo improprio (Banfi *et al.*, 2012).

Exsiccata: Isola di Giannutri, sopra all'eliporto, all'interno dell'area con i generatori elettrici; 16.07.2011; *G. Ferretti* & *B. Foggi*; UTM: 32T 673589.4680158 (FI). Monte Argentario (GR), Porto S. Stefano, ex proprietà Jacovacci, bordi della strada; 03.07.1994; *R.M. Baldini* (FI! sub *Datura inoxia*).

! ***Drosanthemum floribundum*** (Haw.) Schwantes (Aizoaceae)
(≡ *Mesembryanthemum floribundum* Haw.)
[CAS] - **Elba**: CAS; **Giglio**: CAS

Prima segnalazione per la Toscana. In Arcipelago è presente all'Elba (Pomonte, parte alta del paese) e al Giglio (Giglio Campese). Si tratta comunque di stazioni sporadiche, molto vincolate alla presenza di individui coltivati.

Exsiccata: Isola d'Elba, Pomonte, parte alta del paese, margine stradale; 28.09.2012; *G. Ferretti* & *C. Giuliani*; UTM: 32T 591909.4733595 (FI). Isola del Giglio, Giglio Campese, ai margini della Torre del Campese; 15.07.2011; *G. Ferretti* & *B. Foggi*; UTM: 32T 654879.4692443 (FI).

* ***Dysphania ambrosioides*** (L.) Mosyakin & Clemants (Chenopodiaceae)
(≡ *Chenopodium ambrosioides* L. = *Chenopodium suffruticosum* Willd.)
[NAT] - **Elba**: NAT

Prima segnalazione per l'Arcipelago Toscano, dove è stata rilevata all'Isola d'Elba in località Seccheto, S. Andrea e San Mamiliano presso Marina di Campo. Sulle coste mediterranee si trovano spesso esemplari con base legnosa, a volte indicati come *Chenopodium suffruticosum*, che tuttavia rientrano nell'ampia variabilità della specie (Clemants & Mosyakin, 2003).

Exsiccata: Isola d'Elba, Campo nell'Elba, Seccheto, spiaggia; 20.06.2008; *G. Galasso*; UTM: 32T 596348.4732194 (MSNM).

* ***Ipomoea indica*** (Burm.) Merr. (Convolvulaceae)
(≡ *Convolvulus indicus* L.)
[NAT] - **Elba**: NAT

Prima segnalazione per l'Arcipelago Toscano, dove è molto utilizzata come pianta ornamentale; l'abbiamo osservata naturalizzata all'Isola d'Elba, a nordest di Porto Azzurro. Citata per la Toscana da Celesti-Grapow *et al.* (2009, 2010).

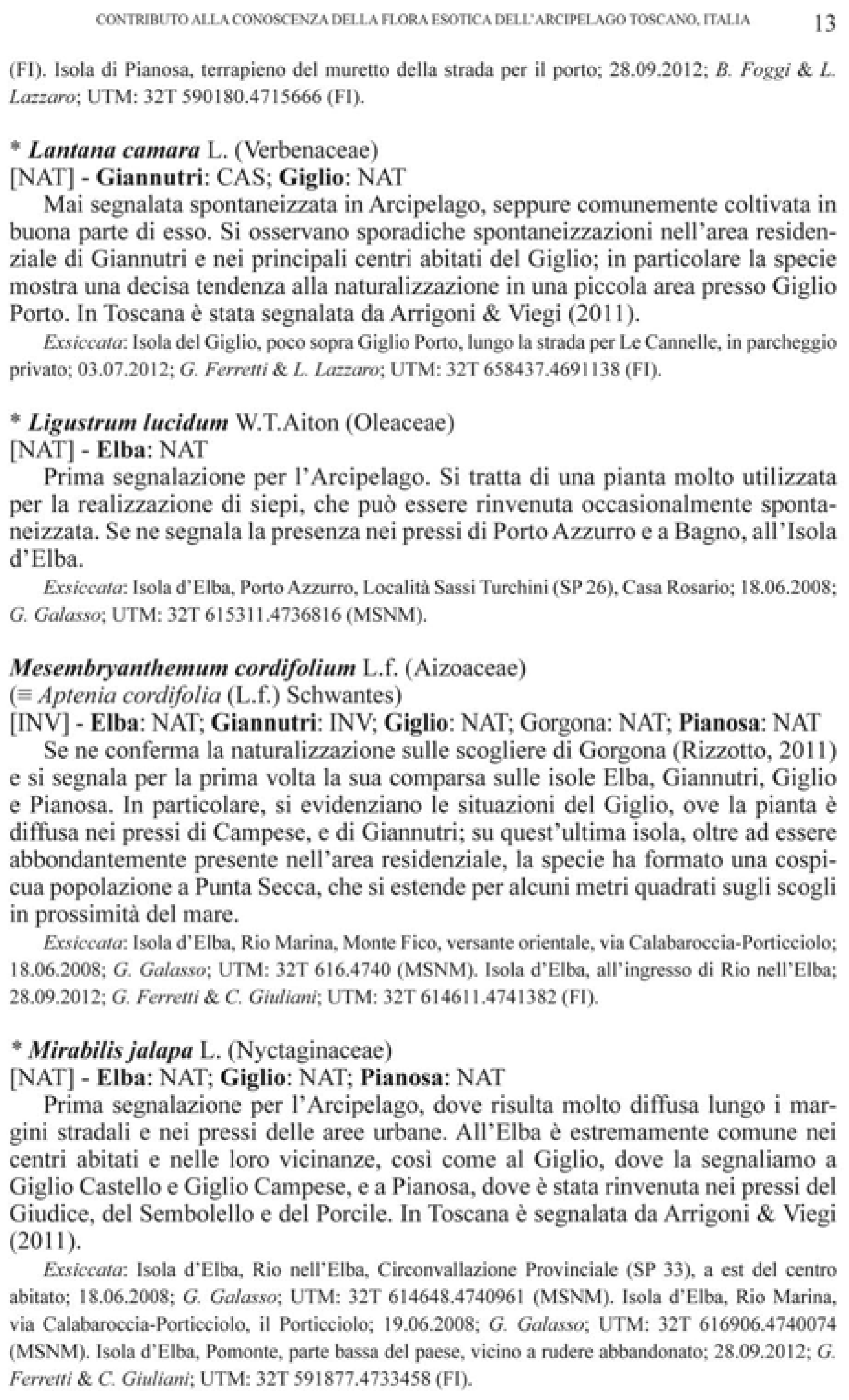
Exsiccata: Isola d'Elba, Porto Azzurro, via Provinciale est (SP 26); 19.06.2008; *G. Galasso*; UTM: 32T 614704.4736534 (MSNM).

! ***Kalanchoë ×houghtonii*** D.B.Ward (Crassulaceae)
(≡ *Bryophyllum ×houghtonii* (D.B.Ward) P.I.Forst. = *Kalanchoë daigremontiana* auct., non Raym.-Hamet & H.Perrier = *Kalanchoë daigremontiana* Raym.-Hamet & H.Perrier × *Kalanchoë delagoensis* Eckl. & Zeyh.)
[NAT] - **Elba**: CAS; **Giannutri**: NAT; **Pianosa**: NAT

Ibrido originatosi in serra a partire da specie (allopatriche) malgascie, descritto di recente (Ward, 2006) ma già conosciuto da prima come pianta esotica invasiva, spesso confuso con *K. daigremontiana*. La pianta può essere distinta agevolmente in base alla forma della base della lamina e alla tipologia della dentatura (Shaw, 2008; Ward, 2008); il colore dei fiori è invece variabile, andando dal rosso scuro citato nel protologo (Ward, 2006) fino al salmone (Hannan-Jones & Playford, 2002).

Questi ritrovamenti costituiscono la seconda segnalazione per l'Italia (Podda *et al.*, 2012) e la prima per la Toscana, ma è sicuramente più diffuso in quanto prima non veniva distinto da *K. daigremontiana*. Ad esempio, il dato relativo a *K. daigremontiana* citato per la Toscana da Celesti-Grapow *et al.* (2009, 2010) è da riferire a questo ibrido in quanto deriva dal campione di Rio nell'Elba. Nella maggior parte dei casi (Elba, Rio Marina; Giannutri, in paese; Pianosa, in paese) le popolazioni rinvenute sono ampie e rilevanti, anche se al momento restano circoscritte alle aree antropizzate. Largamente naturalizzato in Australia (Hannan-Jones & Playford, 2002; Hosking *et al.*, 2003), Caraibi e Florida (Ward, 2006, 2008), si diffonde principalmente (se non esclusivamente) per via vegetativa. Nell'Arcipelago le nuove piante non sembrano in grado di distanziarsi molto dai nuclei di origine, anche se a Giannutri iniziano a collocarsi sempre più spesso ai margini della macchia mediterranea. Il maggior rischio per le nuove colonizzazioni risiede nell'abbandono degli scarti di pulizia dei giardini.

Exsiccata: Isola d'Elba, Rio nell'Elba, Circonvallazione Provinciale (SP 33), a est del centro abitato; 18.06.2008; *G. Galasso*; UTM: 32T 614648.4740961 (MSNM). Isola di Giannutri, nei pressi di Cala Spalmatoio, margini stradali; 16.07.2011; *G. Ferretti & B. Foggi*; UTM: 32T 673585.4680221



(FI). Isola di Pianosa, terrapieno del muretto della strada per il porto; 28.09.2012; *B. Foggi* & *L. Lazzaro*; UTM: 32T 590180.4715666 (FI).

* ***Lantana camara*** L. (Verbenaceae)
[NAT] - **Giannutri**: CAS; **Giglio**: NAT

Mai segnalata spontaneizzata in Arcipelago, seppure comunemente coltivata in buona parte di esso. Si osservano sporadiche spontaneizzazioni nell'area residenziale di Giannutri e nei principali centri abitati del Giglio; in particolare la specie mostra una decisa tendenza alla naturalizzazione in una piccola area presso Giglio Porto. In Toscana è stata segnalata da Arrigoni & Viegi (2011).

Exsiccata: Isola del Giglio, poco sopra Giglio Porto, lungo la strada per Le Cannelle, in parcheggio privato; 03.07.2012; *G. Ferretti* & *L. Lazzaro*; UTM: 32T 658437.4691138 (FI).

* ***Ligustrum lucidum*** W.T.Aiton (Oleaceae)
[NAT] - **Elba**: NAT

Prima segnalazione per l'Arcipelago. Si tratta di una pianta molto utilizzata per la realizzazione di siepi, che può essere rinvenuta occasionalmente spontaneizzata. Se ne segnala la presenza nei pressi di Porto Azzurro e a Bagno, all'Isola d'Elba.

Exsiccata: Isola d'Elba, Porto Azzurro, Località Sassi Turchini (SP 26), Casa Rosario; 18.06.2008; *G. Galasso*; UTM: 32T 615311.4736816 (MSNM).

Mesembryanthemum cordifolium L.f. (Aizoaceae)
(≡ *Aptenia cordifolia* (L.f.) Schwantes)
[INV] - **Elba**: NAT; **Giannutri**: INV; **Giglio**: NAT; Gorgona: NAT; **Pianosa**: NAT

Se ne conferma la naturalizzazione sulle scogliere di Gorgona (Rizzotto, 2011) e si segnala per la prima volta la sua comparsa sulle isole Elba, Giannutri, Giglio e Pianosa. In particolare, si evidenziano le situazioni del Giglio, ove la pianta è diffusa nei pressi di Campese, e di Giannutri; su quest'ultima isola, oltre ad essere abbondantemente presente nell'area residenziale, la specie ha formato una cospicua popolazione a Punta Secca, che si estende per alcuni metri quadrati sugli scogli in prossimità del mare.

Exsiccata: Isola d'Elba, Rio Marina, Monte Fico, versante orientale, via Calabaroccia-Porticciolo; 18.06.2008; *G. Galasso*; UTM: 32T 616.4740 (MSNM). Isola d'Elba, all'ingresso di Rio nell'Elba; 28.09.2012; *G. Ferretti* & *C. Giuliani*; UTM: 32T 614611.4741382 (FI).

* ***Mirabilis jalapa*** L. (Nyctaginaceae)
[NAT] - **Elba**: NAT; **Giglio**: NAT; **Pianosa**: NAT

Prima segnalazione per l'Arcipelago, dove risulta molto diffusa lungo i margini stradali e nei pressi delle aree urbane. All'Elba è estremamente comune nei centri abitati e nelle loro vicinanze, così come al Giglio, dove la segnaliamo a Giglio Castello e Giglio Campese, e a Pianosa, dove è stata rinvenuta nei pressi del Giudice, del Sembolello e del Porcile. In Toscana è segnalata da Arrigoni & Viegi (2011).

Exsiccata: Isola d'Elba, Rio nell'Elba, Circonvallazione Provinciale (SP 33), a est del centro abitato; 18.06.2008; *G. Galasso*; UTM: 32T 614648.4740961 (MSNM). Isola d'Elba, Rio Marina, via Calabaroccia-Porticciolo, il Porticciolo; 19.06.2008; *G. Galasso*; UTM: 32T 616906.4740074 (MSNM). Isola d'Elba, Pomonte, parte bassa del paese, vicino a rudere abbandonato; 28.09.2012; *G. Ferretti* & *C. Giuliani*; UTM: 32T 591877.4733458 (FI).

! ***Myoporum insulare*** R.Br. (Scrophulariaceae)

[CAS] - **Elba**: CAS; **Giannutri**: CAS

Prima segnalazione per la Toscana. La specie, identificata tramite le chiavi di Chinnock (2007), è stata rinvenuta all'Isola d'Elba (Patresi) e all'Isola di Giannutri (Cala Spalmatoio). In entrambi i casi si tratta di introduzioni effettuate a fini colturali, che occasionalmente mostrano tendenza alla spontaneizzazione. Ciò è particolarmente evidente a Giannutri, ove alcuni individui si sono stabiliti sulle scogliere in prossimità del mare.

Riportiamo inoltre la presenza di *Myoporum laetum* G.Forst. sull'Isola di Montecristo, dove era stata segnalata da Paoli & Romagnoli (1976, sub *M. punctulatum* Schltdl.). Trattandosi di un solo individuo, evidentemente coltivato, non riteniamo però opportuno inserirla nella lista delle specie spontaneizzate.

Exsiccata: (*Myoporum insulare*) Isola d'Elba, Patresi, margine stradale lungo la SP 25, in prossimità del paese (Marciana); 28.05.2012; *G. Ferretti & B. Foggi*; UTM: 32T 591478.4738406 (FI). Isola di Giannutri, Cala Spalmatoio, sugli scogli presso il porto; 16.07.2011; *G. Ferretti & B. Foggi*; UTM: 32T 673607.4680150 (FI). (*Myoporum laetum*) Isola di Montecristo, Cala Maestra, nei pressi dell'abitato; 14.05.2011; *G. Ferretti*; UTM: 32T 606826.4687667 (FI).

Opuntia monacantha Haw. (Cactaceae) (Figg. 6-7)

[NAT] - Capraia: NR; Elba: NAT; **Giannutri**: NAT; **Montecristo**: NAT; **Pianosa**: CAS

Si confermano le informazioni di Guiggi (2008) relative sia alla presenza sull'Isola d'Elba sia al mancato ritrovamento della stazione di Capraia (*exsiccata sine data et legit* in TO). Risultano nuove invece le segnalazioni per le isole di Giannutri, Montecristo e Pianosa. Si tratta di pochi esemplari spontaneizzati nei pressi delle aree abitate, dove le piante coltivate generano nuovi individui che tendono a mantenersi nelle loro vicinanze.

Fig. 6 - Isola d'Elba: *Opuntia monacantha* lungo la strada per il Monserrato. / Elba Island: *Opuntia monacantha* along the road to Monserrato. (Foto / Photo L. Lazzaro).

Fig. 7 - Isola di Pianosa: *Opuntia monacantha* nei pressi dell'abitato. / Pianosa Island: *Opuntia monacantha* near the village. (Foto / Photo G. Rizzo).

Opuntia stricta (Haw.) Haw. (Cactaceae) (Figg. 8-9)
(≡ *Cactus strictus* Haw.)
[INV] - Capraia: INV; **Giglio**: INV; Pianosa: NR

Il campione raccolto da Sommier nel 1901 presso il Cardon di Pianosa, conservato in FI e attribuito a *O. ficus-indica* (L.) Mill. è stato rideterminato da Guiggi (2008) come *O. stricta*. Tuttavia questa segnalazione non risulta confermata dalle recenti indagini svolte sull'isola. La specie è stata invece confermata a Capraia, dove era già stata riportata da Guiggi (2008) e dove risulta estremamente abbondante nelle garighe e sulle scogliere in vicinanza del porto e del paese. Infine al Giglio, qui riportata per la prima volta, risulta ben rappresentata sia lungo il margine della strada per Capel Rosso che nei pressi delle Cannelle; in quest'ultima località mostra tutta la sua capacità invasiva occupando ampie superfici sia nelle garighe che su rupi e scogliere.

Oxalis articulata Savigny (Oxalidaceae)
(= *Oxalis violacea* auct., non L.)
[NAT] - Capraia: NAT; **Elba**: NAT; Giglio: NAT; Gorgona: CAS; **Montecristo**: NAT; **Pianosa**: NAT

Se ne conferma la presenza sulle isole di Capraia (Foggi *et al.*, 2001), Giglio (Baldini, 1998) e Gorgona (Rizzotto, 2011). Risultano invece nuove le segnalazioni per Elba, Pianosa e Montecristo, dove *O. articulata* è abbastanza ben rappresentata in vicinanza dei centri abitati. Oltre ai campioni citati, all'Elba risulta presente in numerose località tra cui Acquaviva, Marciana Marina, Marina di Campo, Poggio e Pomonte, mentre a Pianosa è stata osservata presso la Cala dei Turchi e La Centrale. A Montecristo è piuttosto comune nei pressi dell'area abitativa, dove sfugge alla coltivazione e si naturalizza anche sulle sponde del Rio

Fig. 8 - Isola del Giglio: *Opuntia stricta* su rupi e in gariga nei pressi della frazione Le Cannelle. / Giglio Island: *Opuntia stricta* on rocks and in garigue near the village Le Cannelle. (Foto / Photo L. Lazzaro).

Fig. 9 - Isola del Giglio: *Opuntia stricta* su rupi e in gariga a Le Cannelle. / Giglio Island: *Opuntia stricta* on rocks and in garigue at Le Cannelle. (Foto / Photo L. Lazzaro).

di Cala Maestra. Va precisato che, nell'ambito dell'Arcipelago Toscano, la specie è stata raccolta per la prima volta sull'Isola di Montecristo da Fabbri, Contardo e Bavazzano nel 1965 e successivamente segnalata da Paoli & Romagnoli (1976), i quali tuttavia l'avevano erroneamente attribuita a *O. violacea*. Una recente revisione del materiale conservato in FI da parte di Adriano Stinca ha corretto la precedente determinazione; *O. violacea* risulta pertanto entità da eliminare dalla flora dell'Arcipelago Toscano.

Exsiccata: Isola d'Elba, Capoliveri, via Capo ai Pini (SP 30), Casa Riccio; 18.06.2008; *G. Galasso*; UTM: 32T 609069.4735497 (MSNM). Isola d'Elba, Portoferraio, via Valle di Lazzaro (SP 24), tra Carpani e Campitelle; 19.06.2008; *G. Galasso*; UTM: 32T 606609.4739223 (MSNM). Isola di Montecristo, presso l'abitato; 06.05.2012; *G. Ferretti*; UTM: 32T 606742.4687622 (FI).

* ***Oxalis debilis*** Kunth (Oxalidaceae)
(= *Oxalis corymbosa* DC. ≡ *O. debilis* Kunth subsp. *corymbosa* (DC.) O.Bolòs & Vigo ≡ *O. debilis* Kunth var. *corymbosa* (DC.) Lourteig)
[NAT] - **Elba**: NAT

Prima segnalazione per l'Arcipelago Toscano, ove è stata ritrovata all'Elba, negli acciottolati del centro storico di San Piero in Campo. Per la sinonimia con *O. corymbosa* si veda Banfi & Galasso (2012). In Toscana è stata riportata per il viareggino da Arrigoni & Viegi (2011).

Exsiccata: Isola d'Elba, Campo nell'Elba, San Piero in Campo, piazza della chiesa; 22.06.2008; *G. Galasso*; UTM: 32T 599126.4733946 (MSNM).

Oxalis dillenii Jacq. (Oxalidaceae)
[NAT] - **Elba**: NAT; Giannutri: NAT

Prima segnalazione per l'Isola d'Elba, dove è stata rinvenuta a Sant'Ilario in Campo e presso San Piero in Campo; citata per Giannutri da Baldini (2001). Specie comune, a volte confusa con *Oxalis stricta* L. (= *O. fontana* Bunge) (Galasso, 2009), spesso naturalizzata in ambiente antropico; segnalata in Toscana da Conti *et al.* (2007).

Exsiccata: Isola d'Elba, Campo nell'Elba, via della Costa (SP 25), C. Vecchia; 20.06.2008; *G. Galasso*; UTM: 32T 592958.4732293 (MSNM). Isola d'Elba, Campo nell'Elba, Sant'Ilario in Campo, nucleo storico; 22.06.2008; *G. Galasso*; UTM: 32T 5993.47353 (MSNM).

* ***Oxalis latifolia*** Kunth (Oxalidaceae) (Figg. 10-11)
[NAT] - **Elba**: NAT; **Giglio**: NAT

Prima segnalazione per l'Arcipelago, dove se ne rileva la presenza all'isola d'Elba, presso Pomonte e Marina di Campo, e al Giglio, presso Campese. La specie è stata rinvenuta sia all'interno di aiuole stradali che in vasi e giardini. Anche se al momento appare localizzata, la rapidità con cui *O. latifolia* sta occupando nuovi spazi nella Toscana continentale (Arrigoni & Viegi, 2011; Pierini, 2011) suggerisce una sua probabile espansione anche in Arcipelago.

Paraserianthes lophantha (Willd.) I.C.Nielsen (Fabaceae)
(≡ *Acacia lophantha* Willd. ≡ *Albizia lophantha* (Willd.) Benth.)
[NAT] - **Capraia**: NAT; Elba: NAT

Prima segnalazione per l'isola di Capraia, dove il nucleo principale è costituito da diversi individui, abbondantemente fruttificanti, che hanno colonizzato un'area abbandonata dislocata nei pressi del Faro. Ai margini del paese sono state osservate

Fig. 10 - Isola del Giglio: *Oxalis latifolia* nelle aiuole stradali di Giglio Campese. / Giglio Island: *Oxalis latifolia* in the road lawns of Giglio Campese. (Foto / Photo L. Lazzaro).

Fig. 11 - Isola del Giglio: *Oxalis latifolia* nelle aiuole stradali di Giglio Campese. / Giglio Island: *Oxalis latifolia* in the road lawns of Giglio Campese. (Foto / Photo L. Lazzaro).

altre piccole stazioni, costituite da uno o pochi individui sviluppatisi tra le macerie o negli incolti. Anche per Capraia si conferma lo status di naturalizzata già assegnato per l'Isola d'Elba, dove la specie è stata segnalata per la prima volta in Toscana (Frangini *et al.*, 2010). Riteniamo opportuno in questa sede citare un'ulteriore stazione per l'Elba, lungo la strada che da Capoliveri conduce al Monte Calamita, particolarmente interessante in quanto posta a notevole distanza da centri abitati e strade.

Exsiccata: Isola di Capraia, in un'area abbandonata al limite del paese, non distante dal Faro; 06.05.2011; *L. Lastrucci, B. Foggi* & *R. Calamassi*; UTM: 32T 568666.4766697 (FI); Isola di Capraia, Capraia Isola, incolto presso la Torre del Porto; 06.2011; *B. Foggi, L. Lastrucci* & *R. Calamassi*; UTM: 32T 0568.4766 (MSNM).

* ***Parthenocissus quinquefolia*** (L.) Planch. (Vitaceae)
(≡ *Hedera quinquefolia* L. = *Vitis inserta* A.Kern. ≡ *Parthenocissus inserta* (A.Kern.) Fritsch)
[INV] - **Elba**: INV; **Giannutri**: INV; **Giglio**: INV; **Pianosa**: NAT

Prima segnalazione per l'Arcipelago Toscano, dove è sicuramente presente da lungo tempo come pianta coltivata a scopo ornamentale. In questa sede se ne conferma l'elevata invasività e la si indica come pianta spontaneizzata estremamente abbondante in corrispondenza di buona parte dei centri abitati di Elba, Giannutri, Giglio e Pianosa, dove tende a stabilirsi anche al margine dei boschi, penetrando talora al loro interno. La ricca produzione di frutti carnosi, abbondantemente mangiati e dispersi dagli uccelli, la rende una delle specie invasive con maggiori capacità di diffusione. Segnaliamo anche i suoi potenziali danni alle infrastrutture, particolarmente evidenti quando la pianta si addossa ai manufatti, ricoprendoli completamente. In Toscana è citata da Arrigoni & Viegi (2011).

Exsiccata: Isola d'Elba, Portoferraio, in periferia, lungo la SP 24; 28.09.2012; *G. Ferretti* & *C. Giuliani*; UTM: 32T 606856.4739598 (FI). Isola di Pianosa, paese, nei pressi della casa dell'Agronomo; 28.09.2012; *B. Foggi* & *L. Lazzaro*; UTM: 32T 590067.4715544 (FI).

* ***Passiflora caerulea*** L. (Passifloraceae)
[NAT] - **Elba**: NAT

Prima segnalazione per l'Arcipelago Toscano; coltivata con una certa frequenza, riesce a spontaneizzare con discreta facilità. Generalmente tende a mantenersi nelle vicinanze delle piante madri, ma sempre più spesso se ne rileva la presenza in situazioni svincolate dal contesto antropico. All'Isola d'Elba sono stati accertati individui spontanei in località Seccione, a Punta della Crocetta presso Marciana Marina e lungo la strada tra Porto Azzurro e il Monserrato. In Toscana è indicata da Arrigoni & Viegi (2011).

Exsiccata: Isola d'Elba, Porto Azzurro, strada per il Monserrato; 28.09.2012; *G. Ferretti* & *C. Giuliani*; UTM: 32T 614665.4736977 (FI).

* ***Pittosporum tobira*** (Thunb.) W.T.Aiton (Pittosporaceae)
(≡ *Euonymus tobira* Thunb.)
[NAT] - **Capraia**: CAS; **Elba**: NAT; **Giannutri**: CAS; **Giglio**: CAS

Prima segnalazione per l'Arcipelago Toscano. Comunemente coltivata per la realizzazione di siepi, tende a spontaneizzare sia in vicinanza dei centri abitati sia in ambiti più naturali, quali macchie, garighe e boschi radi. All'Elba presenta numerose stazioni sparse su tutto il territorio, con individui giovani riscontrabili anche all'interno della macchia mediterranea; più sporadica la sua presenza a Capraia (tra il Porto e il Paese), Giglio (Campese) e Giannutri (Cala Spalmatoio). In Toscana è segnalata da Arrigoni & Viegi (2011).

! ***Tetragonia tetragonoides*** (Pall.) Kuntze (Aizoaceae)
(≡ *Demidovia tetragonoides* Pall. ≡ *Tetragonia expansa* Murray, nom. illeg.)
[CAS] - **Giannutri**: CAS

Prima segnalazione per la Toscana. In FI sono stati rinvenuti due campioni d'erbario, mai citati in letteratura, provenienti da Viareggio e da Marina di Pisa. Pur essendo nota e coltivata a scopo alimentare già nei primi anni del '900, non è mai stata segnalata spontaneizzata. In Arcipelago l'abbiamo rinvenuta sull'Isola di Giannutri, presso Cala Maestra.

Exsiccata: Isola di Giannutri, presso il vecchio porto romano di Cala Maestra, nella spianata a ciottoli; 21.06.2012; *G. Ferretti & L. Lazzaro*; UTM: 32T 672916.4680121 (FI). Viareggio, in hortis colitus; 20.09.1913; *G. Gemmi* (FI!). Marina di Pisa: arena di spiaggia alla foce dell'Arno; 18.08.1925; *M. Savelli* (FI!).

Tropaeolum majus L. (Tropaeolaceae)
[CAS] - Capraia: CAS; **Elba**: CAS

Prima segnalazione per l'Isola d'Elba, recentemente segnalata per l'Isola di Capraia da Lastrucci *et al*. (2012); coltivata per ornamento, occasionalmente la si può rinvenire spontaneizzata. Già citata per la Toscana da Celesti-Grapow *et al.* (2009, 2010) e Arrigoni & Viegi (2011).

Exsiccata: Isola d'Elba, Rio Marina, via Calabaroccia-Porticciolo, il Porticciolo; 18.06.2008; *G. Galasso*; UTM: 32T 616906.4740074 (MSNM).

! ***Vachellia karroo*** (Hayne) Banfi & Galasso (Fabaceae)
(≡ *Acacia karroo* Hayne)
[CAS] - **Elba**: CAS

Prima segnalazione per la Toscana, all'Isola d'Elba, dove è stata trovata una piccola stazione in località Campitelle (Portoferraio), costituita da alcuni individui nati da seme sul terrapieno di un muro a secco.

Exsiccata: Isola d'Elba, loc. Campitelle, strada SP 24 per Marciana presso il bivio per San Martino (Portoferraio); 17.02.2012; *G. Ferretti & L. Lazzaro*; UTM: 32T 605835.4738909 (FI).

Ziziphus jujuba Mill. (Rhamnaceae)
(≡ *Rhamnus zizyphus* L. ≡ *Ziziphus zizyphus* (L.) H.Karst., nom. rej.)
[NAT] - Elba: NAT; **Pianosa**: NAT

La specie è naturalizzata a Pianosa dove è presente una cospicua popolazione al Pollaio. Nel territorio elbano era stata segnalata da Thiébaut De Berneaud (1808), non confermata da Fossi Innamorati (1994) e riportata da Arrigoni & Viegi (2011); recentemente ne abbiamo rinvenuto numerosi individui naturalizzati su una rupe a bordo strada al di sotto dell'abitato di San Pietro in Campo. Si ricorda che di recente il nome *Z. jujuba* è stato conservato rispetto a *Z. zizyphus* (Paclt, 1999; Brummitt, 2000; Kirkbride *et al.*, 2006; Brummitt, 2009).

Exsiccata: Isola di Pianosa, viottolo interno del Pollaio; 29.09.2012; *L. Lazzaro & B. Foggi*; UTM: 32T 589424.4715281 (FI).

Conclusioni

Il presente contributo costituisce un ulteriore passo in avanti verso la compilazione di un lavoro organico sulla spontaneizzazione delle specie esotiche in Arcipelago Toscano. Le piccole isole spesso ospitano comunità vegetali che si sono evolute in un ambiente sottoposto a basse spinte competitive, per cui risultano particolarmente sensibili alle invasioni biologiche (Pretto *et al.*, 2010), giungendo a ospitare una maggior proporzione di specie aliene rispetto all'entroterra (Pyšek & Richardson, 2006). È questo il caso dell'Arcipelago Toscano, dove il numeroso contingente esotico esercita il suo impatto su superfici limitate e spesso di alto valore conservazionistico.

La maggior parte delle specie qui riportate sono state introdotte come piante ornamentali e successivamente si sono spontaneizzate in aree marginali non curate dall'uomo. Risulta difficile risalire al periodo della loro prima introduzione in Arcipelago, anche se, verosimilmente, alcune potrebbero essere presenti da lungo tempo, pur se mai segnalate allo stato spontaneo. Molte specie alloctone necessitano di un lungo periodo di adattamento alle nuove condizioni ecologiche ed è logico supporre che, superato questo *lag time*, alcune fra le numerose introduzioni recenti potranno costituire un serio problema (Pretto *et al.*, 2010). Le continue introduzioni di esotiche aumentano le potenziali sorgenti di neo-diffusione dei propaguli (*propagule pressure*), aumentando notevolmente la probabilità di eventi invasivi o di naturalizzazioni (Pretto *et al.*, 2012); oltretutto molte specie ornamentali vengono immesse sul mercato in seguito a un processo di selezione che le rende maggiormente adattabili al nuovo ambiente (Lambdon *et al.*, 2008; Pretto *et al.*, 2012) e questo può aumentare ulteriormente le possibilità di spontaneizzazione. Tutti questi fattori evidenziano quanto sia importante mantenere alto il grado di attenzione nel monitoraggio continuo delle spontaneizzazioni, anche in ambienti molto antropizzati e anche per specie ornamentali molto conosciute, ma il più delle volte sottovalutate dal punto di vista del loro potenziale impatto ambientale.

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3.2 Second contribution to the knowledge of the alien flora of Tuscan Archipelago, Italy.

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Secondo contributo alla conoscenza della flora esotica dell'Arcipelago Toscano, Italia

Riassunto - Si presenta qui un secondo contributo per la realizzazione di una *Checklist* della flora esotica dell'Arcipelago Toscano. Per ogni *taxon* sono riportati lo *status* di naturalizzazione, sia per ciascuna isola che per l'intero Arcipelago, e i dati relativi alla distribuzione locale. Delle 31 specie elencate, 18 risultano nuove per il territorio dell'Arcipelago Toscano, tra le quali 4 nuove per l'intera Toscana; le restanti sono nuove per una o più isole.

Parole chiave: Arcipelago Toscano, flora esotica, Mediterraneo, status.

Abstract - Second contribution to the knowledge of the alien flora of the Tuscan Archipelago, Italy.

A second update to the realization of the Check list of the alien flora in the Tuscan Archipelago is presented. For each *taxon* the local distribution and the naturalization status are specified. 18 of the 31 species listed herewith were not previously recorded for the Tuscan Archipelago, and 4 of these are new to Tuscan flora. The remaining ones are new to different islands.

Keywords: alien flora, Mediterranean, status, Tuscan Archipelago.

Introduzione

Il presente lavoro si colloca nell'ambito del progetto di realizzazione di una *Checklist* delle piante esotiche presenti allo stato spontaneo nell'Arcipelago Toscano (Fig. 1). Un primo contributo sull'argomento è stato recentemente pubblicato da Lazzaro *et al.* (2013). Si presentano qui ulteriori aggiornamenti consistenti in una serie di nuove segnalazioni per le singole isole o per l'intero Arcipelago, alcune delle quali risultano inedite per la Toscana. A queste si aggiungono alcune conferme per specie segnalate precedentemente al 1950 e prima d'ora non riconfermate. Tutte queste informazioni incrementano le conoscenze floristiche sul contingente esotico presente in Toscana.

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Materiali e metodi

Le note floristiche di seguito riportate derivano da osservazioni effettuate in campagna nel corso degli anni 2012-2013. Le località di rinvenimento sono state georeferenziate tramite GPS e, salvo rare eccezioni, si è sempre proceduto al prelievo di materiale da erbario. Gli *exsiccata* sono stati depositati presso l'Erbario del Museo di Storia Naturale di Milano (MSNM).

La nomenclatura utilizzata segue Banfi & Galasso (2010) e Celesti-Grapow *et al.* (2010), salvo diversa indicazione. Le specie sono riportate in ordine alfabetico e, laddove necessario, sono indicati il basionimo e i principali sinonimi. Per ogni entità sono fornite le informazioni su presenza, distribuzione e *status* di naturalizzazione nelle singole isole; per la valutazione di quest'ultimo sono state adottate le definizioni di Richardson *et al.* (2000) e Pyšek *et al.* (2004): casuale, naturalizzato, invasivo. In particolare la corretta attribuzione degli status si è basata su informazioni inedite raccolte dagli autori nell'ultimo decennio, dati bibliografici pregressi e infine osservazioni effettuate direttamente sulle popolazioni individuate nell'area di studio. Anteposto al binomio specifico ed in parentesi quadra è indicato lo *status* in Arcipelago (considerando lo stadio più avanzato del processo di invasione tra quelli delle singole isole, secondo l'ordine invasivo > naturalizzato >

Fig. 1 - Inquadramento geografico dell'Arcipelago Toscano. / Geographical overview of Tuscan Archipelago.

casuale); segue l'elenco alfabetico delle isole sulle quali la specie è stata rinvenuta allo stato spontaneo, ciascuna con il grado di naturalizzazione presentato su di essa dalla specie in questione. Si evidenziano anche segnalazioni precedenti al 1950 e mai riconfermate. I dati sulla presenza in Toscana sono desunti da Viegi & Cela Renzoni (1981), Conti *et al.* (2005, 2007), Celesti-Grapow *et al.* (2009, 2010), Arrigoni & Viegi (2011), oltre che da specifiche pubblicazioni riguardanti i singoli territori insulari, evidenziate di volta in volta.

Nel testo viene utilizzata la seguente simbologia:
! = prima segnalazione per la Toscana,
* = prima segnalazione per l'Arcipelago Toscano,
isola in grassetto = prima segnalazione per l'isola,
isola in corsivo = conferma di segnalazione antica (precedente al 1950),
CAS = casuale,
NAT = naturalizzata,
INV = invasiva,
NR = specie non ritrovata dopo il 1950, presumibilmente estinta localmente.

Laddove siano stati raccolti campioni si cita la dicitura del cartellino d'erbario, nel seguente ordine: località; data di raccolta; raccoglitore/i; coordinate cartografiche nel sistema di riferimento UTM (WGS84) ed erbario in cui è stato depositato il saggio. Il termine *observata* si riferisce invece a quelle segnalazioni prive di un campione d'erbario per le quali si forniscono comunque dati stazionali completi.

Risultati e discussione

Amaranthus albus L. (Amaranthaceae)
[NAT] - *Capraia*: NAT; *Elba*: NAT; Giglio: NR; Gorgona: NAT; Montecristo: NR; Pianosa: NR

Specie segnalata fino ai primi del '900 su buona parte del territorio dell'Arcipelago Toscano: Capraia, Elba, Giglio, Montecristo e Pianosa (Requien, 1852; Sommier, 1903, 1909). Non era più stata rinvenuta su nessuna delle isole con la sola eccezione di Gorgona, dove è stata segnalata da Rizzotto (2011). Recenti ritrovamenti nei pressi di Marina di Campo ne confermano la presenza per l'Isola d'Elba, mentre sull'Isola di Capraia sono stati trovati abbondanti popolamenti nell'area del Piano di S. Stefano.

Exsiccata: Isola d'Elba, Marina di Campo, area incolta alla periferia del paese; 11.10.2012; G. Ferretti & C. Giuliani; UTM: 32T 601105.4734238 (MSNM). Isola di Capraia, loc. Piano di Santo Stefano, all'interno del vigneto e ai margini di questo; 12.9.2013; G. Ferretti & L. Lazzaro; UTM: 32T 567544.4765055 (MSNM).

* ***Amaranthus caudatus*** L. (Amaranthaceae)
[CAS] - **Elba**: CAS

Specie segnalata, prima del 1950, nelle province di Pistoia e Firenze (Viegi & Cela Renzoni, 1981). Di recente confermata per il Monte Pisano (Pierini *et al.*, 2009) dove è nota dalla fine degli anni '80 (Del Prete *et al.*, 1991); nell'Arcipelago non era mai stata rinvenuta. Viene qui segnalata la sua presenza nell'abitato di Pomonte dove casualmente sfugge alla coltivazione come pianta da ornamento, insediandosi lungo i margini stradali.

Exsiccata: Isola d'Elba, Pomonte, margine stradale del paese; 11.10.2012; *G. Ferretti & C. Giuliani*; UTM: 32T 591954.4733485 (MSNM).

Antirrhinum majus L. subsp. ***majus*** (Plantaginaceae)
[NAT] - Elba: NAT; **Giglio**: NAT; Pianosa: NAT

Entità piuttosto comune nella Toscana continentale, già segnalata in arcipelago sulle isole di Pianosa (Baldini, 2000) ed Elba (Sommier, 1903; Rinaldi, 2002). Se ne è da poco accertata la presenza anche sull'isola del Giglio dove *Antirrhinum majus* subsp. *majus* è naturalizzata su alcuni muri a secco nei dintorni di Giglio Castello.

Exsiccata: Isola del Giglio, Giglio Castello, uscendo dal paese in direzione di Capel Rosso; 03.07.2012; *G. Ferretti & L. Lazzaro*; UTM: 32T 656555.4691900 (MSNM). Isola d'Elba, Marina di Campo, loc. Pozzalmoro, in prossimità di un fosso; 11.10.2012; *G. Ferretti & C. Giuliani*; UTM: 32T 600055.4732968 (MSNM).

Broussonetia papyrifera (L.) Vent. (Moraceae)
(≡ *Morus papyrifera* L.)
[NAT] - Capraia: CAS; **Pianosa**: NAT

Specie già nota in arcipelago per l'isola di Capraia, dove era stata segnalata per la prima volta da Sommier (1898) e confermata da Foggi *et al.* (2001), come specie casuale ai bordi stradali. A Pianosa se ne è riscontrata la presenza di alcuni individui, naturalizzati, in un'area ruderale nei pressi dell'attuale ristorante.

Exsiccata: Isola di Pianosa, paese, area ruderale nel retro del ristorante; 09.10.2012; *G. Ferretti & L. Lazzaro*; UTM: 32T 589957.4715618 (MSNM).

* ***Canna indica*** L. (Cannaceae)
[CAS] - **Elba**: CAS

Specie segnalata recentemente in Toscana lungo il litorale nord-occidentale (Marchetti, 2011). Se ne evidenzia la presenza all'Isola d'Elba, in due stazioni: la prima tra Rio Marina e Rio Elba, lungo un fosso adiacente alla strada, l'altra tra Marciana Marina e Poggio, nei pressi della località Timonaia.

Exsiccata: Isola d'Elba, strada tra Rio Marina e Rio Elba, nel fosso a bordo strada; 10.10.2012; *G. Ferretti & C. Giuliani*; UTM: 32T 615534.4740924 (MSNM).

Observata: Isola d'Elba, tra Marciana Marina e Poggio, loc. Timonaia; 11.10.2012; *G. Ferretti & C. Giuliani*; UTM: 32T 597304.4738512.

! ***Catharanthus roseus*** (L.) G. Don (Apocynacae)
(≡ *Vinca rosea* L.)
[NAT] - **Elba**: NAT

Entità già segnalata sul territorio nazionale per la sola Calabria (Celesti-Grapow *et al.*, 2010). In Toscana, pur essendo largamente coltivata, non era mai stata evidenziata in situazioni di spontaneizzazione. All'Isola d'Elba, nell'interno del paese di Pomonte, *Catharanthus roseus* è riuscita ad affermarsi andando a costituire piccoli, ma numerosi nuclei naturalizzati, posti ai margini di strade o sentieri, nel selciato stradale o sui muri in pietra.

Exsiccata: Isola d'Elba, Pomonte, in paese; 11.10.2012; *G. Ferretti & C. Giuliani*; UTM: 32T 591954.4733485 (MSNM).

* ***Chamaesyce humifusa*** (Willd. ex Schltr.) Prokh. (Euphorbiaceae)
(≡ *Euphorbia humifusa* Willd. ex Schltr.)
[NAT] - **Elba**: NAT

Exsiccata: Isola d'Elba, Pomonte, aiuole e marciapiedi nella piazza della chiesa; 11.10.2012; *G. Ferretti & C. Giuliani*; UTM: 32T 591776.4733518 (MSNM).

Chamaesyce maculata (L.) Small (Euphorbiaceae)
(≡ *Euphorbia maculata* L.)
[NAT] - Capraia: NAT; **Elba**: NAT; **Pianosa**: NAT

Segnalata a Capraia sulla base di un *exsiccatum* di Bavazzano del 1969 (Viegi & Cela Renzoni, 1981). Se ne segnala la presenza anche all'Isola d'Elba, a Pomonte, e all'isola di Pianosa, dove è stata rinvenuta in varie aree a carattere ruderale, quali Il Giudice, Il Panificio e nei pressi dell'attuale ristorante.

Exsiccata: Isola d'Elba, Pomonte, aiuole e marciapiedi nella piazza della chiesa; 11.10.2012; *G. Ferretti & C. Giuliani*; UTM: 32T 591776.4733518 (MSNM). Pianosa, presso Il Giudice; 08.10.2012; *G. Ferretti & L. Lazzaro*; UTM: 32T 589072.4714496 (MSNM).

Observata: Pianosa, presso il Panificio; 08.10.2012; *G. Ferretti & L. Lazzaro*; UTM: 32T 589806.4715611. Pianosa, area ruderale nel retro dell'attuale ristorante; 08.10.2012; *G. Ferretti & L. Lazzaro*; UTM: 32T 589957.4715618.

Chamaesyce prostrata (Aiton) Smal. (Euphorbiaceae)
(≡ *Euphorbia prostrata* Aiton)
[NAT] - Giannutri: NAT; **Giglio**: NAT

Già segnalata solo per l'Isola di Giannutri (Baldini, 2000), ne è ora accertata la presenza anche all'Isola del Giglio, al Campese.

Exsiccata: Isola del Giglio, Campese, nei pressi del campo sportivo; 03.07.2012; *G. Ferretti & L. Lazzaro*; UTM: 32T 654808.4691891 (MSNM).

Citrullus lanatus (Thunb.) Matsum. & Nakai (Cucurbitaceae) (Fig. 2)
(≡ *Momordica lanata* Thunb. = *Citrullus vulgaris* Schrad.)
[CAS] - *Elba*: CAS; **Giannutri**: CAS; *Gorgona*: CAS

Entità già segnalata per l'Elba da Thiébaut De Berneaud (1808, sub *Cucumis anguria*) nella piana di Porto Azzurro e riportata come coltivata da Fossi Innamorati (1994); è tuttora coltivata nell'isola dove occasionalmente si spontaneizza. Un'antica segnalazione di Arcangeli (1888, sub *Citrullus vulgaris* Schrd.) per Gorgona, non confermata da Rizzotto (2011), viene qui convalidata. La pianta è stata ritrovata allo stato spontaneo vicino alla piccola spiaggia di Gorgona Scalo. Se ne segnala, infine, la presenza casuale anche sull'Isola di Giannutri, dove è stata osservata sugli scogli di Cala Spalmatoio.

Observata: Isola d'Elba, al margine stradale della Via Provinciale Ovest, vicino al bivio per Capoliveri; 11.10.2012; *G. Ferretti & C. Giuliani*; UTM: 32T 611965.4735126. Isola di Giannutri, scogli di Cala Spalmatoio; 22.06.2012; *G. Ferretti & L. Lazzaro*; UTM: 32T 673605.4680150. Isola di Gorgona, Gorgona Scalo presso la spiaggia vicino al porto; 08.11.2012; *G. Ferretti, C. Giuliani & L. Lazzaro*; UTM: 32T 573360.4809113.

* ***Cucurbita maxima*** Duchesne (Cucurbitaceae)
[CAS] - **Elba**: CAS; **Gorgona**: CAS

Occasionalmente spontaneizzata in varie località della Toscana continentale, come ad esempio nelle campagne di Pistoia, Prato e Lucca (GF, osservazione personale), ma non ancora segnalata nell'Arcipelago. È stata rinvenuta di recente all'Isola d'Elba, nel paese di Pomonte, e a Gorgona, nei pressi dell'area attualmente destinata a discarica di rifiuti.

Exsiccata: Isola d'Elba, Pomonte, al margine della strada che scende al mare; 11.10.2012; *G. Ferretti & C. Giuliani*; UTM: 32T 591650.4733469 (MSNM).

Observata: Isola di Gorgona, area della discarica; 08.11.2012; *G. Ferretti, C. Giuliani & L. Lazzaro*; UTM: 32T 572534.4809137.

Fig. 2 - Isola di Gorgona: *Citrullus lanatus* presso la spiaggia vicino al porto (Foto G. Ferretti). / Gorgona Island: *Citrullus lanatus* at the beach close to the harbor (Photo G. Ferretti).

* ***Cyperus involucratus*** Rottb. (Cyperaceae)
[NAT] - **Elba**: NAT

Exsiccata: Isola d'Elba, periferia di Portoferraio, lungo un fosso al margine della SP 24; 04.02.2013; *L. Lazzaro*; UTM: 32T 606989.4739951 (MSNM).

* ***Datura ferox*** L. (Solanaceae)
[CAS] - **Pianosa**: CAS

La presenza di questa specie in Toscana è documentata da un vecchio reperto raccolto nel 1891 a Pisa (Viegi & Cela Renzoni, 1981). In seguito non è più stata segnalata nella regione tanto che Celesti-Grapow (2010) e Arrigoni & Viegi (2011) la considerano assente.

È stata ritrovata a Pianosa in due stazioni, la prima all'interno di un'area ruderale nel retro dell'attuale ristorante e l'altra presso il vecchio Panificio.

Exsiccata: Isola di Pianosa, paese, area ruderale nel retro del ristorante; 09.10.2012; *G. Ferretti & L. Lazzaro*; UTM: 32T 589957.4715618 (MSNM).

Observata: Isola di Pianosa, paese, presso il vecchio Panificio; 10.10.2012; *L. Lazzaro*; UTM: 32T 589701.4715617.

Datura stramonium L. subsp. ***stramonium*** (Solanaceae)
[NAT] - Capraia: NAT; Elba: NAT; Giglio: NR; Gorgona: NAT; Montecristo: NR; **Pianosa**: NAT

Segnalata a Capraia, sull'isolotto della Peraiola dove secondo Foggi *et al.* (2001) è giunta dalla fine degli anni '90 a seguito della presenza dei gabbiani. Evidenziata all'Isola d'Elba già dalla seconda metà dell'ottocento e raccolta più di recente, presso Mola, da Fossi Innamorati nel 1975 (Fossi Innamorati, 1989; exs in FI). Confermata a Gorgona da Rizzotto (2011) e invece non più rinvenuta a Giglio e Montecristo.

Si riporta qui il suo primo ritrovamento sull'Isola di Pianosa dove è stata individuata in numerose aree con popolamenti naturalizzati e in fase di espansione.

Exsiccata: Isola di Pianosa, presso il Giudice; 08.10.2012; *G. Ferretti & L. Lazzaro*; UTM: 32T 589034.4714475 (MSNM).

Observata: Isola di Pianosa, presso il Porcile, negli ex coltivi attorno alla casa; 08.10.2012; *G. Ferretti & L. Lazzaro*; UTM: 32T 589248.4713879. Isola di Pianosa, presso il Pollaio; 08.10.2012; *G. Ferretti & L. Lazzaro*; UTM: 32T 589500.4715347. Isola di Pianosa, nei pressi dell'acquedotto; 08.10.2012; *G. Ferretti & L. Lazzaro*; UTM: 32T 588684.4714954. Isola di Pianosa, nei dintorni del faro; 08.10.2012; *G. Ferretti & L. Lazzaro*; UTM: 32T 589961.4715398. Isola di Pianosa, paese, area ruderale nel retro del ristorante; 09.10.2012; *G. Ferretti & L. Lazzaro*; UTM: 32T 589957.4715618.

* ***Eleusine indica*** (L.) Gaertn. subsp. ***indica*** (Poaceae)
(≡ *Cynosurus indicus* L.)
[NAT] - **Pianosa**: NAT

Entità estremamente frequente nella Toscana continentale, di cui non era mai stata evidenziata la presenza in arcipelago. È stata rinvenuta a Pianosa, presso il Pollaio, all'interno di un piccolo orto abbandonato.

Exsiccata: Isola di Pianosa, presso Il Pollaio, in orto abbandonato; 08.10.2012; *G. Ferretti & L. Lazzaro*; UTM: 32T 589500.4715347 (MSNM).

* ***Erigeron sumatrensis*** Retz. (Asteraceae)
(≡ *Conyza sumatrensis* (Retz.) E. Walker = *C. albida* Spreng. = *C. naudinii* Bonnet – *C. floribunda* auct. non Kunth)
[NAT] - **Elba**: NAT; **Giannutri**: NAT; **Giglio**: NAT; **Gorgona**: NAT; **Pianosa**: NAT; **Montecristo**: NAT

La presenza di tale specie non era finora stata evidenziata su nessuna delle isole dell'Arcipelago Toscano. Recenti indagini di campagna, nonché la revisione del materiale d'erbario conservato in FI, hanno permesso invece di attestarne una certa diffusione su alcune delle maggiori isole (Elba, Giglio, Gorgona). La specie è sempre legata a contesti antropici dove appare ben affermata e talora in espansione. La sua mancata documentazione è probabilmente da imputare al recente arrivo in Arcipelago Toscano. La revisione del materiale in FI ha permesso di constatare l'assenza di exsiccata precedenti il 1960. In seguito i primi ritrovamenti sono stati spesso confusi con le due entità affini (*E. bonariensis* L. ed *E. canadensis* L.). Le tre specie infatti convivono, anche su superfici molto circoscritte, in buona parte della Toscana continentale.

Exsiccata: Isola d'Elba, Cavo, lungo le strade del paese; 10.10.2012; *G. Ferretti & C. Giuliani*; UTM: 32T 615882.4746234 (MSNM). Isola d'Elba, Fosso Mandriola; 29.08.1973; *T. Fossi Innamorati*; (FI! sub *Conyza canadensis*). Isola d'Elba, Nella macchia dietro il villaggio di S. Michele di Bagnaia; 03.09.1972; *T. Fossi Innamorati*; (FI! sub *Conyza canadensis*). Isola del Giglio, lungo la strada per l'Arenella; 03.07.2012; *G. Ferretti & L. Lazzaro*; UTM: 32T 657233.4692494 (MSNM). Isola di Gorgona, nei pressi del "Vigneto", in loc. Il Campone; 08.11.2012; *G. Ferretti, C. Giuliani & L. Lazzaro*; UTM: 32T 572995.4809250 (MSNM). Isola di Pianosa, presso il Porcile; 08.10.2012; *G. Ferretti & L. Lazzaro*; UTM: 32T 589202.4713868 (MSNM).

! ***Erythrostemon gilliesii*** (Wall. ex Hook.) Klotzsch (Fabaceae) (Figg. 3-4)
(≡ *Poinciana gilliesii* Hook. ≡ *Caesalpinia gilliesii* (Hook.) D. Dietr.)
[CAS] - **Elba**: CAS

In Celesti-Grapow *et al*. (2010) è ritenuta naturalizzata in Sicilia e casuale in Sardegna. Finora nota in Toscana solamente allo stato coltivato. Ne è stata accertata la sua spontaneizzazione, anche se del tutto casuale, all'Isola d'Elba. È presente nel paese di Pomonte all'interno di un vecchio giardino abbandonato.

Observata: Isola d'Elba, Pomonte, spontanea all'interno di un giardino abbandonato; 11.10.2012; *G. Ferretti & C. Giuliani*; UTM: 32T 591874.4733476.

* ***Fallopia baldschuanica*** (Regel) Holub (Polygonaceae)
(≡ *Polygonum baldschuanicum* Regel)
[NAT] - **Elba**: NAT

Specie abbastanza frequente nella Toscana continentale, ma non ancora segnalata per l'Arcipelago Toscano. È stata ritrovata con una certa frequenza all'Isola d'Elba, sia nei pressi dell'abitato di Poggio che lungo la strada tra Poggio e Marciana Marina.

Exsiccata: Isola d'Elba, strada tra Marciana Marina e Poggio, loc. Timonaia; 11.10.2012; *G. Ferretti & C. Giuliani*; UTM: 32T 597262.4738627 (MSNM).

Fig. 3 - Isola d'Elba: *Erythrostemon gilliesii* nell'abitato di Pomonte, spontanea all'interno di un giardino abbandonato (Foto G. Ferretti). / Elba Island: *Erythrostemon gilliesii* at the village of Pomonte, growing in a abandoned backyard (Photo G. Ferretti).

Fig. 4 - Isola d'Elba: *Erythrostemon gilliesii*, particolare della pianta nell'abitato di Pomonte (Foto G. Ferretti). / Elba Island: *Erythrostemon gilliesii* detail of the plant at Pomonte village (Photo G. Ferretti).

* ***Gleditsia triacanthos*** L. (Fabaceae)
[CAS] - **Elba**: CAS

Segnalata in numerose località toscane, ma non ancora rinvenuta sulle isole dell'arcipelago. È stata ritrovata con presenza occasionale all'Isola d'Elba, lungo la strada per il Volterraio, nei pressi della località Schiopparello.

Exsiccata: Isola d'Elba, strada per il Volterraio, loc. Schiopparello, margine stradale; 10.10.2012; *G. Ferretti & C. Giuliani*; UTM: 32T 611383.4739449 (MSNM).

Helianthus tuberosus L. (Asteraceae)
[NAT] - **Elba**: NAT; Gorgona: CAS

La prima e unica segnalazione di questa specie nell'Arcipelago Toscano si riferisce all'Isola di Gorgona, dove la pianta è nota dagli anni '90 (Gori, 1993; Rizzotto, 2011).

È stata di recente rinvenuta all'Isola d'Elba, dove risulta ben affermata in numerose stazioni distribuite su tutto il territorio: Campitelle, Zanca, Porto Azzurro, Portoferraio, Patresi, Marina di Campo.

Exsiccata: Isola d'Elba, lungo la SP "Anello occidentale", presso il bivio per Zanca.; 11.10.2012; *G. Ferretti & C. Giuliani*; UTM: 32T 592610.4739106 (MSNM).

Mirabilis jalapa L. (Nyctaginaceae)
[NAT] - Elba: NAT; Giglio: NAT; **Gorgona**: NAT; Pianosa: NAT

Specie segnalata di recente nell'arcipelago sulle isole di Pianosa, Giglio ed Elba (Lazzaro *et al*., 2013). Da poco rinvenuta naturalizzata anche a Gorgona Scalo.

Exsiccata: Isola di Gorgona, Gorgona Scalo, nei pressi del porto; 08.11.2012; *G. Ferretti, C. Giuliani & L. Lazzaro*; UTM: 32T 573415.4809192 (MSNM).

! *Nothoscordum gracile* (Aiton) Stearn (Alliaceae)
(≡ *Allium gracile* Aiton = *A. fragrans* Vent. – *Nothoscordum inodorum* auct. plur., non (Aiton) Nicholson)
[CAS] - **Pianosa**: CAS

Entità dal trattamento nomenclaturale a lungo dibattuto ma chiarito nel lavoro di Stearn (1986) a cui qui ci si riferisce. Già nota per diverse regioni italiane: Liguria (Schiappacasse & Peccenini, 2009), Lazio e Campania (Celesti-Grapow *et al*., 2010, sub *N. borbonicum* Kunth), Sicilia (Raimondo *et al*., 2010), Sardegna (Bacchetta *et al.*, 2009). In Toscana la sua presenza non era finora stata rilevata.

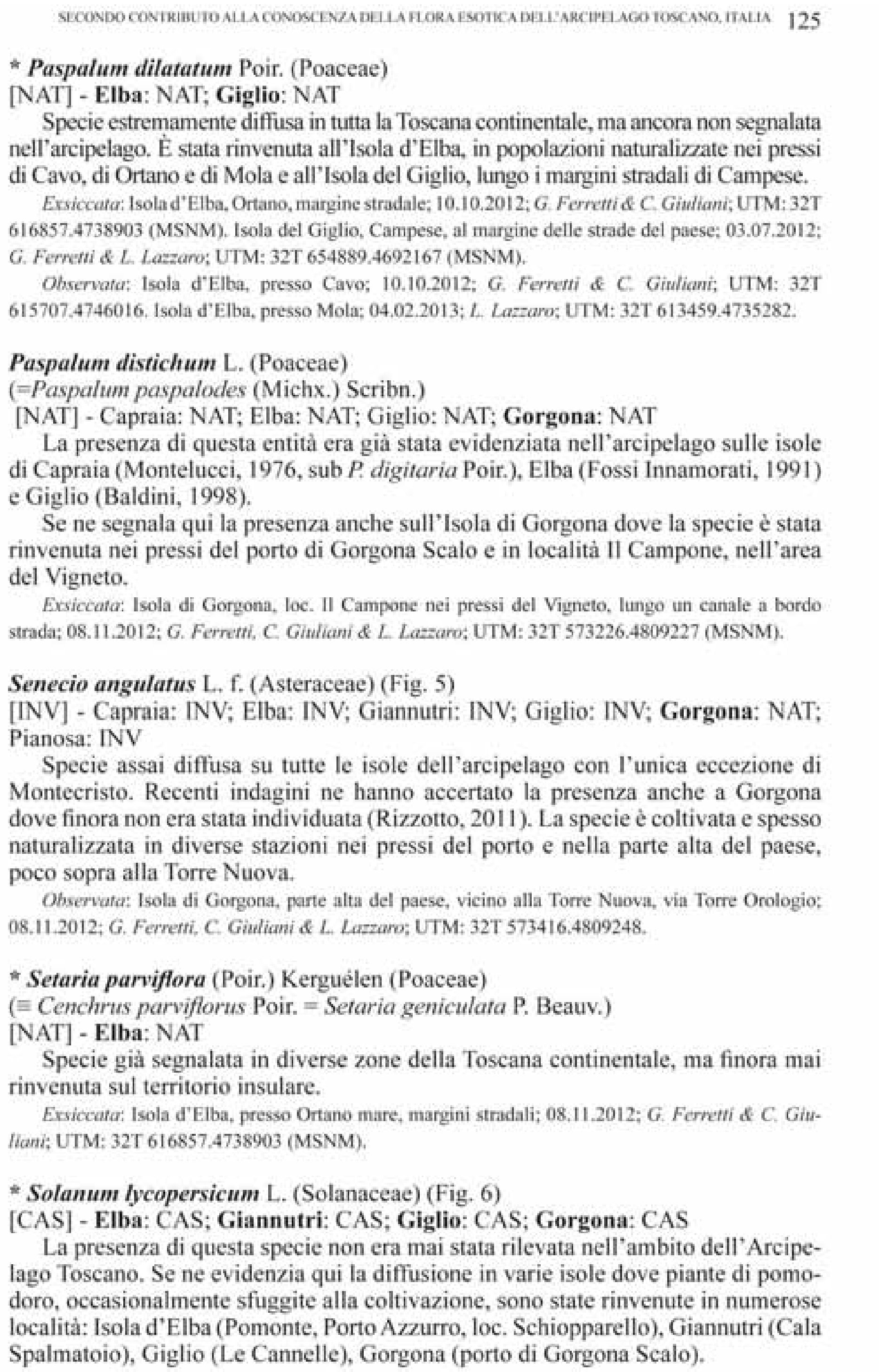
Exsiccata: Isola di Pianosa, presso Il Pollaio, in orto abbandonato; 08.10.2012; *G. Ferretti & L. Lazzaro*; UTM: 32T 589500.4715347 (MSNM).

! *Oxalis bowiei* Herb. ex Lindl. (Oxalidaceae)
(≡ *Oxalis purpurata* Jacq. var. *bowiei* (Herb. ex Lindl.) Sond.)
[NAT] - **Elba**: NAT

Prima segnalazione per la Toscana, dove è stata rinvenuta all'Isola d'Elba, nei pressi di Mola.

Entità probabilmente introdotta a scopo ornamentale in giardini privati e quindi diffusa, in condizioni di chiara spontaneizzazione, negli incolti e sulla scarpata a lato della Via Provinciale Ovest. Ad oggi risulta assente dal territorio nazionale, sebbene nota in Campania nella prima metà del 1900, ma non ritrovata successivamente. *O. bowiei* risulta sistematicamente affine a *O. purpurata* ampiamente segnalata in Italia da Celesti-Grapow *et al*. (2010) (LOM, LIG, TOS, LAZ, ABR, SIC) e dalla quale si distingue per alcuni caratteri riguardanti la grandezza del bulbo e delle foglie (Muñoz Garmendia & Navarro, 2013), il colore e la dimensione dei fiori (Jacquin, 1798; Sonder, 1859-60). Seguendo il trattamento di Flora Iberica (Muñoz Garmendia & Navarro, 2013) *Oxalis bowiei* viene qui riconosciuta come *taxon* indipendente da *O. purpurata*. Dai campioni d'erbario conservati in FI, *O. bowiei* risulta introdotta in Toscana già a fine '800 come specie coltivata, sia presso il giardino di acclimatazione della Casa Bianca di Porto Ercole che nell'Orto Botanico di Firenze (dove è tutt'ora in coltivazione). Un *exsiccatum* in FI ne testimonia la coltivazione anche in Liguria (presso Varazze) fin dai primi del '900. C'è inoltre da evidenziare che la segnalazione di Baldini (1995) di *O. purpurata* per l'Argentario è da riferire a *O. articulata* Savigny (*exsiccatum* in FI, *revidit* A. Stinca), mentre quella di Selvi (2010) per la maremma grossetana è priva di campione di riferimento e necessita di conferma (Selvi *in verbis*). La nostra segnalazione deve quindi essere considerata la prima per la Toscana e la conferma della sua presenza allo stato spontaneo per il territorio nazionale.

Exsiccata: Isola d'Elba, Portoazzurro, loc. Mola, campi e margine stradale.; 10.10.2012; *G. Ferretti & C. Giuliani*; UTM: 32T 613262.4735076 (MSNM).



* ***Paspalum dilatatum*** Poir. (Poaceae)
[NAT] - **Elba**: NAT; **Giglio**: NAT

Specie estremamente diffusa in tutta la Toscana continentale, ma ancora non segnalata nell'arcipelago. È stata rinvenuta all'Isola d'Elba, in popolazioni naturalizzate nei pressi di Cavo, di Ortano e di Mola e all'Isola del Giglio, lungo i margini stradali di Campese.

Exsiccata: Isola d'Elba, Ortano, margine stradale; 10.10.2012; *G. Ferretti & C. Giuliani*; UTM: 32T 616857.4738903 (MSNM). Isola del Giglio, Campese, al margine delle strade del paese; 03.07.2012; *G. Ferretti & L. Lazzaro*; UTM: 32T 654889.4692167 (MSNM).

Observata: Isola d'Elba, presso Cavo; 10.10.2012; *G. Ferretti & C. Giuliani*; UTM: 32T 615707.4746016. Isola d'Elba, presso Mola; 04.02.2013; *L. Lazzaro*; UTM: 32T 613459.4735282.

Paspalum distichum L. (Poaceae)
(=*Paspalum paspalodes* (Michx.) Scribn.)
[NAT] - Capraia: NAT; Elba: NAT; Giglio: NAT; **Gorgona**: NAT

La presenza di questa entità era già stata evidenziata nell'arcipelago sulle isole di Capraia (Montelucci, 1976, sub *P. digitaria* Poir.), Elba (Fossi Innamorati, 1991) e Giglio (Baldini, 1998).

Se ne segnala qui la presenza anche sull'Isola di Gorgona dove la specie è stata rinvenuta nei pressi del porto di Gorgona Scalo e in località Il Campone, nell'area del Vigneto.

Exsiccata: Isola di Gorgona, loc. Il Campone nei pressi del Vigneto, lungo un canale a bordo strada; 08.11.2012; *G. Ferretti, C. Giuliani & L. Lazzaro*; UTM: 32T 573226.4809227 (MSNM).

Senecio angulatus L. f. (Asteraceae) (Fig. 5)
[INV] - Capraia: INV; Elba: INV; Giannutri: INV; Giglio: INV; **Gorgona**: NAT; Pianosa: INV

Specie assai diffusa su tutte le isole dell'arcipelago con l'unica eccezione di Montecristo. Recenti indagini ne hanno accertato la presenza anche a Gorgona dove finora non era stata individuata (Rizzotto, 2011). La specie è coltivata e spesso naturalizzata in diverse stazioni nei pressi del porto e nella parte alta del paese, poco sopra alla Torre Nuova.

Observata: Isola di Gorgona, parte alta del paese, vicino alla Torre Nuova, via Torre Orologio; 08.11.2012; *G. Ferretti, C. Giuliani & L. Lazzaro*; UTM: 32T 573416.4809248.

* ***Setaria parviflora*** (Poir.) Kerguélen (Poaceae)
(≡ *Cenchrus parviflorus* Poir. = *Setaria geniculata* P. Beauv.)
[NAT] - **Elba**: NAT

Specie già segnalata in diverse zone della Toscana continentale, ma finora mai rinvenuta sul territorio insulare.

Exsiccata: Isola d'Elba, presso Ortano mare, margini stradali; 08.11.2012; *G. Ferretti & C. Giuliani*; UTM: 32T 616857.4738903 (MSNM).

* ***Solanum lycopersicum*** L. (Solanaceae) (Fig. 6)
[CAS] - **Elba**: CAS; **Giannutri**: CAS; **Giglio**: CAS; **Gorgona**: CAS

La presenza di questa specie non era mai stata rilevata nell'ambito dell'Arcipelago Toscano. Se ne evidenzia qui la diffusione in varie isole dove piante di pomodoro, occasionalmente sfuggite alla coltivazione, sono state rinvenute in numerose località: Isola d'Elba (Pomonte, Porto Azzurro, loc. Schiopparello), Giannutri (Cala Spalmatoio), Giglio (Le Cannelle), Gorgona (porto di Gorgona Scalo).

Fig. 5 - Isola di Gorgona: *Senecio angulatus* nel parte alta del paese (Foto L. Lazzaro). / Gorgona Island: *Senecio angulatus* at the upper town (Photo L. Lazzaro).

Observata: Isola d'Elba, Pomonte, parte alta del paese; *G. Ferretti & C. Giuliani*; UTM: 32T 591953.4733524. Isola d'Elba, Porto Azzurro, lungo la strada per il Monserrato; *G. Ferretti & C. Giuliani*; UTM: 32T 614357.4737267. Isola d'Elba, località Schiopparello, al margine della strada; *G. Ferretti & C. Giuliani*; UTM: 32T 611290.4738946. Isola di Giannutri, Cala Spalmatoio, scogli vicini all'attracco delle navi; *G. Ferretti & L. Lazzaro*; UTM: 32T 673604.4680149. Isola del Giglio, Le Cannelle, area ruderale in vicinanza della spiaggia; *G. Ferretti & L. Lazzaro*; UTM: 32T 658054.4690492. Gorgona, Gorgona Scalo, nei pressi della spiaggia vicino al porto; *G. Ferretti, C. Giuliani & L. Lazzaro;* UTM: 32T 573361.4809115.

Sorghum halepense (L.) Pers. (Poaceae)
(≡ *Holcus halepense* L.)
[NAT] - *Capraia*: NAT; Elba: NAT; Giannutri: CAS; Montecristo: NR; *Pianosa*: NAT

Specie segnalata in buona parte delle isole dell'arcipelago, talvolta con documentazioni abbastanza recenti, in altri casi attraverso vecchi dati non più riconfermati. Attestazioni di presenza posteriori al 1950 sono quelle relative ad Elba (*exsiccatum* del 1972 in Fossi Innamorati, 1991) e Giannutri (Baldini, 2001). A Montecristo, dopo le citazioni di Caruel (1864) e Sommier (1903), la specie non è più stata ritrovata. A Capraia invece dove, dopo la segnalazione di Moris & De Notaris (1839) e il campione d'erbario del 1847 citato in Viegi & Cela Renzoni (1981), la sua presenza non era stata confermata da Foggi *et al*. (2001), si è rinvenuta con una certa frequenza nelle aree ruderali del porto e del paese. *Sorghum*

Fig. 6 - Isola del Giglio: *Solanum lycopersicum* in area ruderale nei pressi della località Le Cannelle (Foto L. Lazzaro). / Giglio Island: *Solanum lycopersicum* in a ruderal area near Le cannelle (Photo L. Lazzaro).

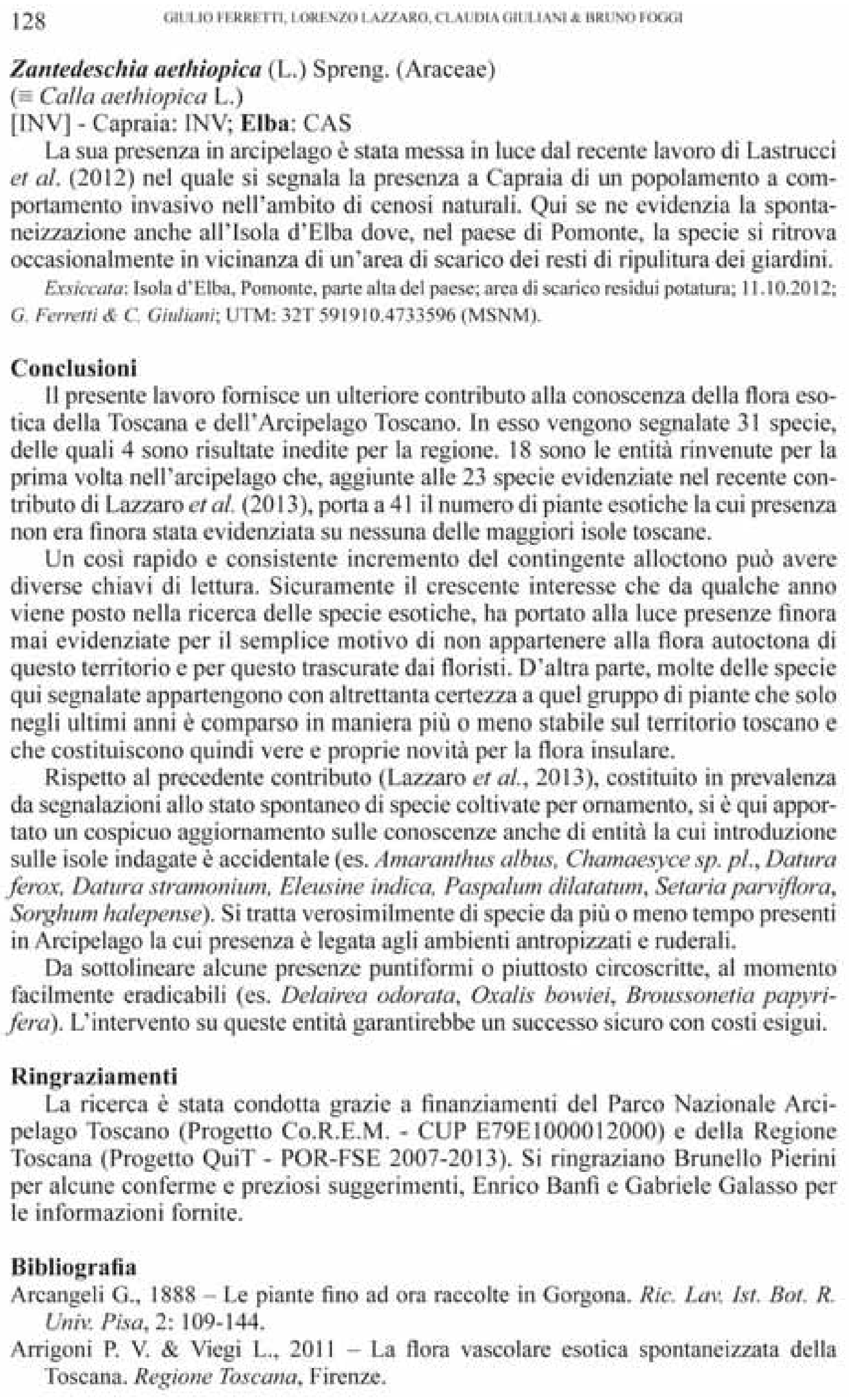
halepense era stato segnalato anche per l'Isola di Pianosa, attraverso citazioni bibliografiche di inizio secolo, sempre riferite ad un reperto di Sommier del 1901. Recenti indagini ne hanno confermato la sua presenza sull'isola lungo la strada sterrata tra il Pollaio e il Giudice.

Exsiccata: Isola di Capraia, aree ruderali nei pressi del porto; 12.9.2013; *G. Ferretti & L. Lazzaro*; UTM: 32T 567961.4766598 (MSNM). Isola di Pianosa, strade tra il frutteto e il Giudice; 08.10.2012; *L. Lazzaro*; UTM: 32T 589683.4714976 (MSNM).

* ***Tradescantia fluminensis*** Vell. (Commelinaceae)
(=*Tradescantia albiflora* Kunth)
[NAT] - **Elba**: NAT

Specie già nota in diverse località della Toscana continentale, in particolare nella Regione Apuana e lungo il versante tirrenico, tra Marina di Carrara e il Monte Pisano (Marchetti, 2011; Peruzzi & Pierini, 2007), ma non ancora segnalata per l'Arcipelago. Il recente ritrovamento, tra Procchio e Marciana Marina, è situato lungo la via delle Mimose in corrispondenza di alcuni cassonetti della spazzatura. Si presume che la spontaneizzazione della pianta abbia preso origine da materiale di potatura rilasciato a bordo strada che si è poi propagato sulle sponde del Fosso Alberelli, dove adesso ricopre alcune decine di metri quadrati.

Exsiccata: Isola d'Elba, tra Procchio e Marciana Marina, lungo la via delle Mimose, sponde del Fosso Alberelli; 11.6.2013; *G. Ferretti & B. Foggi*; UTM: 32T 600295.4737875 (MSNM).



Zantedeschia aethiopica (L.) Spreng. (Araceae)
(≡ *Calla aethiopica* L.)
[INV] - Capraia: INV; **Elba**: CAS

La sua presenza in arcipelago è stata messa in luce dal recente lavoro di Lastrucci *et al*. (2012) nel quale si segnala la presenza a Capraia di un popolamento a comportamento invasivo nell'ambito di cenosi naturali. Qui se ne evidenzia la spontaneizzazione anche all'Isola d'Elba dove, nel paese di Pomonte, la specie si ritrova occasionalmente in vicinanza di un'area di scarico dei resti di ripulitura dei giardini.

Exsiccata: Isola d'Elba, Pomonte, parte alta del paese; area di scarico residui potatura; 11.10.2012; *G. Ferretti & C. Giuliani*; UTM: 32T 591910.4733596 (MSNM).

Conclusioni

Il presente lavoro fornisce un ulteriore contributo alla conoscenza della flora esotica della Toscana e dell'Arcipelago Toscano. In esso vengono segnalate 31 specie, delle quali 4 sono risultate inedite per la regione. 18 sono le entità rinvenute per la prima volta nell'arcipelago che, aggiunte alle 23 specie evidenziate nel recente contributo di Lazzaro *et al.* (2013), porta a 41 il numero di piante esotiche la cui presenza non era finora stata evidenziata su nessuna delle maggiori isole toscane.

Un così rapido e consistente incremento del contingente alloctono può avere diverse chiavi di lettura. Sicuramente il crescente interesse che da qualche anno viene posto nella ricerca delle specie esotiche, ha portato alla luce presenze finora mai evidenziate per il semplice motivo di non appartenere alla flora autoctona di questo territorio e per questo trascurate dai floristi. D'altra parte, molte delle specie qui segnalate appartengono con altrettanta certezza a quel gruppo di piante che solo negli ultimi anni è comparso in maniera più o meno stabile sul territorio toscano e che costituiscono quindi vere e proprie novità per la flora insulare.

Rispetto al precedente contributo (Lazzaro *et al.*, 2013), costituito in prevalenza da segnalazioni allo stato spontaneo di specie coltivate per ornamento, si è qui apportato un cospicuo aggiornamento sulle conoscenze anche di entità la cui introduzione sulle isole indagate è accidentale (es. *Amaranthus albus, Chamaesyce sp. pl.*, *Datura ferox, Datura stramonium, Eleusine indica, Paspalum dilatatum*, *Setaria parviflora*, *Sorghum halepense*). Si tratta verosimilmente di specie da più o meno tempo presenti in Arcipelago la cui presenza è legata agli ambienti antropizzati e ruderali.

Da sottolineare alcune presenze puntiformi o piuttosto circoscritte, al momento facilmente eradicabili (es. *Delairea odorata*, *Oxalis bowiei*, *Broussonetia papyrifera*). L'intervento su queste entità garantirebbe un successo sicuro con costi esigui.

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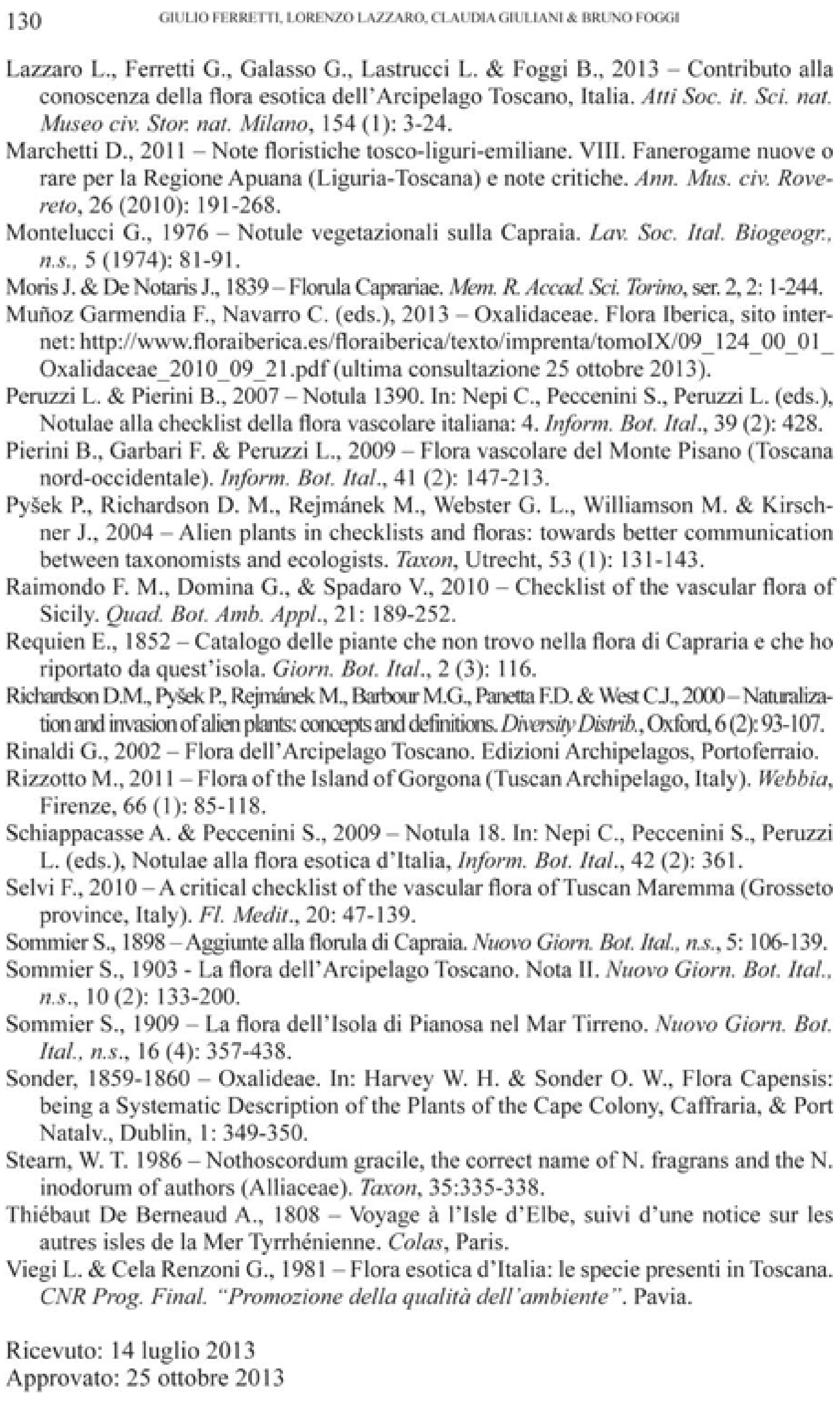

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3.3 A checklist of the alien flora of the Tuscan Archipelago (Italy)

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A checklist of the alien flora of the Tuscan Archipelago (Italy)

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Biological invasions represent one of the major threats to biodiversity. Their impact is considered to be even higher in the island ecosystems, especially in Mediterranean ones, which host a very high diversity. Moreover nowadays the global changes and the local changes in the socio-economic drivers of these island enhanced human impact on the insular biota, also leading to an increase in alien species introductions and in a shift in the introduction pathways. In this framework, checklists of alien species represent a valuable tool for monitor the invasion process at a local scale. Aim of this paper is to provide an updated Checklist of the alien flora of the Tuscan Archipelago, also reporting some important information such year of first and more recent record and invasion status for each island. Intense field surveys were made to verify the presence of the reported species and detect the presence of new arrivals. The Checklist includes a total of 141 taxa belonging to 101 genera in 49 families. 31 taxa are considered invasive, 62 naturalized and 48 casual. Most of the species were introduced for ornamental purpose, according to socio-economical changes which nowadays see the tourism as the main economic driver in the Archipelago.

Keywords: alien flora; conservation; islands; invasions; Mediterranean; Tuscan Archipelago

Introduction

Since the 1990s, the problems caused by invasive alien species have received considerable attention (see Hulme et al. 2013) although the issue of bio(logical)-invasions has a long history, starting from De Candolle's (1855) and Darwin's (1859) first observations.

The Mediterranean basin with more than 12,000 islands and islets (Brundu 2013), hosts one of the largest groups of islands in the world (Montmollin de and Strahm 2005). The high plant diversity of this area is well-known, with about 25,000 native species and a high percentage of endemic species (50–59%: Greuter 1991; Médail and Quèzel 1999). This has led to recognition of the Mediterranean as one of the 34 Global Biodiversity Hotspots (Mittermeier et al. 2004) and its archipelagos could be considered as a 'natural laboratory' for evolutionary studies (Thompson 1999).

According to the future scenario proposed by Sala et al. (2000), Mediterranean ecosystems will be prone to an increase in the invasion rates, especially the islands (Hulme et al. 2007), as the result of changes in important driving factors, e.g. disturbance regimens, land use and climate (Pretto et al. 2010, 2012).

Since the 1960s, the socio-economic status of the Mediterranean islands has changed with a shift from an 'ager-saltus-sylva' meta-stable equilibrium to a new model based on mass tourism (Delanoë et al. 1996). This has enhanced human impact on the insular biota: plants are not introduced for food supply, as in the Neolithic (about 3500 BC) but as ornaments (Hulme 2004).

As stated in the Convention on Biological Diversity decision VI/23 (CBD 2014) the problem of alien invasive species is currently recognized as a focal point for conservation. Strategic Goals for 2020 are that: by 2020, invasive alien species and pathways are identified and prioritized, priority species are controlled or eradicated, and measures are put in place to manage pathways to prevent their introduction and establishment.

According to this general goal, in recent years we started a project devoted to updating the current knowledge of the alien flora of the Tuscan Archipelago (Lastrucci et al. 2012; Lazzaro et al. 2013a; Ferretti et al. 2013). Currently updated Checklists of alien plants of specific areas are considered valuable tools to provide standardized information for such areas, and to evaluate the changes that occur in the invasive status over time, allowing the invasion to be monitored at a local scale (Pyšek et al. 2012).

The aim of this paper is to present a comprehensive and updated Checklist of the alien flora of the Tuscan Archipelago (North Tyrrhenian Sea), which provides as much information as possible on the distribution and invasive status of the alien species for each island and for the whole Archipelago.

Material and Methods

Study area

The Tuscan Archipelago includes seven main islands: Elba, Giglio, Capraia, Montecristo, Pianosa, Giannutri and Gorgona and several islets less than 10 ha in size

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(Figure 1). According to Arrigoni et al. (2003) it represents one of the most interesting areas in the Tyrrhenian Sea from the naturalistic and anthropological perspective. The main geographical information on the Tuscan Archipelago is reported in Table 1. Despite the lack of recent comprehensive works on the flora of the Tuscan Archipelago, the analysis of literature and the exploitation of recent data allow the estimation of spontaneous flora of the Tuscan Archipelago in about 1400 taxa. Of these, 1.2% are strictly endemic, mostly related to Cyrno-Sardinian elements, suggesting that the Tuscan Archipelago represents a bio-geographical bridge between the floristic Cyrno-Sardinian dominion and the Italian peninsula. The landscape of the Tuscan Archipelago is dominated by a typical Mediterranean sclerophyllous–evergreen forest and by its degradation stages (high and low matorrals, garrigues and discontinuous ephemeral grasslands). The human influence on the natural environment started around 4000 BC and has become massive since Roman times (fifth century BC). The strong human presence has deeply impacted the Archipelago's landscape and flora, mainly leading to forest degradation. In the past this was mainly due to agricultural activities, whereas in the last century, like many other Mediterranean islands, the Tuscan Archipelago economy shifted towards tourism activities, with deep changes of its landscape.

Data sources and terminology

The aim of the work is to provide a checklist of the alien flora of the Tuscan Archipelago, considering all the species recorded in at least one of the seven islands after 1950 as casual, naturalized or invasive. Therefore, cryptogenic species and those that were only cultivated were not included. Data collection started with an accurate screening of all the available literature for the Tuscan Archipelago. The screening included all the floras, from the earlier ones (starting from around mid-1850s) towards the most recent, such as the floristic lists of Montecristo (Paoli and Romagnoli 1976) and Elba Island (Fossi Innamorati 1983, 1989, 1991, 1994, 1997), the contributions by Baldini (1998, 2000, 2001) for Giglio, Pianosa and Giannutri and by Foggi et al. (2001) for Capraia, and the literature on invasive alien plants in Tuscany (Viegi and Cela renzoni 1981; Arrigoni and Viegi 2011). The islets' flora (Baldini 1990, 1991; Foggi et al. 2009) and all the recent single contributions and reports were also included.

Figure 1. Geographical overview of the Tuscan Archipelago.

Table 1. Geographical information on the seven main islands of the Tuscan Archipelago.

Island	Surface (Ha)	Altitude (m)	Substrate	Number of spontaneous plants (approx.)	Population (approx.)
Capraia	1931	447	trachites	600 (Foggi et al. 2001; unpubl.)	410
Elba	224,409	1018	granits, trachites, limestone, metamorphic	1250 (Fossi Innamorati 1983, 1989, 1991, 1994, 1997; unpubl.)	33,000
Giannutri	239	93	limestone/panchina	350 (Baldini 2001; unpubl.)	15
Giglio	2154	498	granits/limestome	700 (Baldini 1998; unpubl.)	1500
Gorgona	226	255	metamorphic	500 (Rizzotto 2011; unpubl.)	150
Montecristo	1043	645	granits	400 (Paoli and Romagnoli 1976; unpubl.)	2
Pianosa	1028	27	limestone/panchina	550 (Baldini 2000; unpubl.)	10

Notes: Data on number of spontaneous plant species are approximated according to recent unpublished data.

All of these sources were critically reviewed with the aim to exclude the records of only-cultivated species and to retrieve the precise date (year) of the records. At this stage also, the records relating to wrong identifications were taken into account but solved by checking the herbarium specimens.

The literature review results were integrated with field surveys in all the seven main islands, aiming to update the oldest reports. The islands were surveyed more than once in different period in the years 2011–2013 to verify the presence and persistence of the alien species, to detect new species and to assess the invasion status (see below).

Nomenclature follows Celesti-Grapow et al. (2009) and Banfi and Galasso (2010) and has been further cross-checked using on-line databases (The Plant List 2013; WCSP 2013; Tropicos 2014). These databases, together with specific taxonomic literature, were also used to clarify many of the incongruences in nomenclature found in literature. The names of the families of Angiosperms are after Angiosperm Phylogeny Group III (APG 2009). We followed Blackburn et al. (2011), Celesti-Grapow et al. (2009, 2010a, 2010b) and Pyšek et al. (2004a) for the definition of the alien status of the plant species, considering non-native (and hence 'alien') those species which due to intentional or unintentional human involvement were able to overcome the geographical barriers and be retrieved in the wild in the Tuscan Archipelago. For the *invasion status* terminology we followed the above-mentioned authors in the definition of casual, naturalized and invasive species. We considered as *not retrieved* (NR) the species that were reported in the literature but not retrieved in the wild after 1950.

The species were also divided adopting the widely used distinction (Pyšek et al. 2004b) for residence time, i.e. between archaeophytes (introduced in Europe before the discovery of America in 1492, usually rounded to 1500), and neophytes (introduced after that date). We followed Celesti-Grapow et al. (2009; 2010a; 2010b) and the literature to assess the residence status.

Information concerning the Raunkiaer's life forms (Raunkiaer 1934) was obtained from Pignatti (1982), Cullen et al. (1986/2000), from on-line databases such as JSTOR (2000/2014) and from field observations. The information about the introduction pathway was collected by checking the literature and on-line databases; we follow a simplified classification, distinguishing between voluntary introduction (specifying the main introduction purpose, e.g. forestry, ornamental or agriculture) and accidental introduction (unintentional).

Results

List of taxa

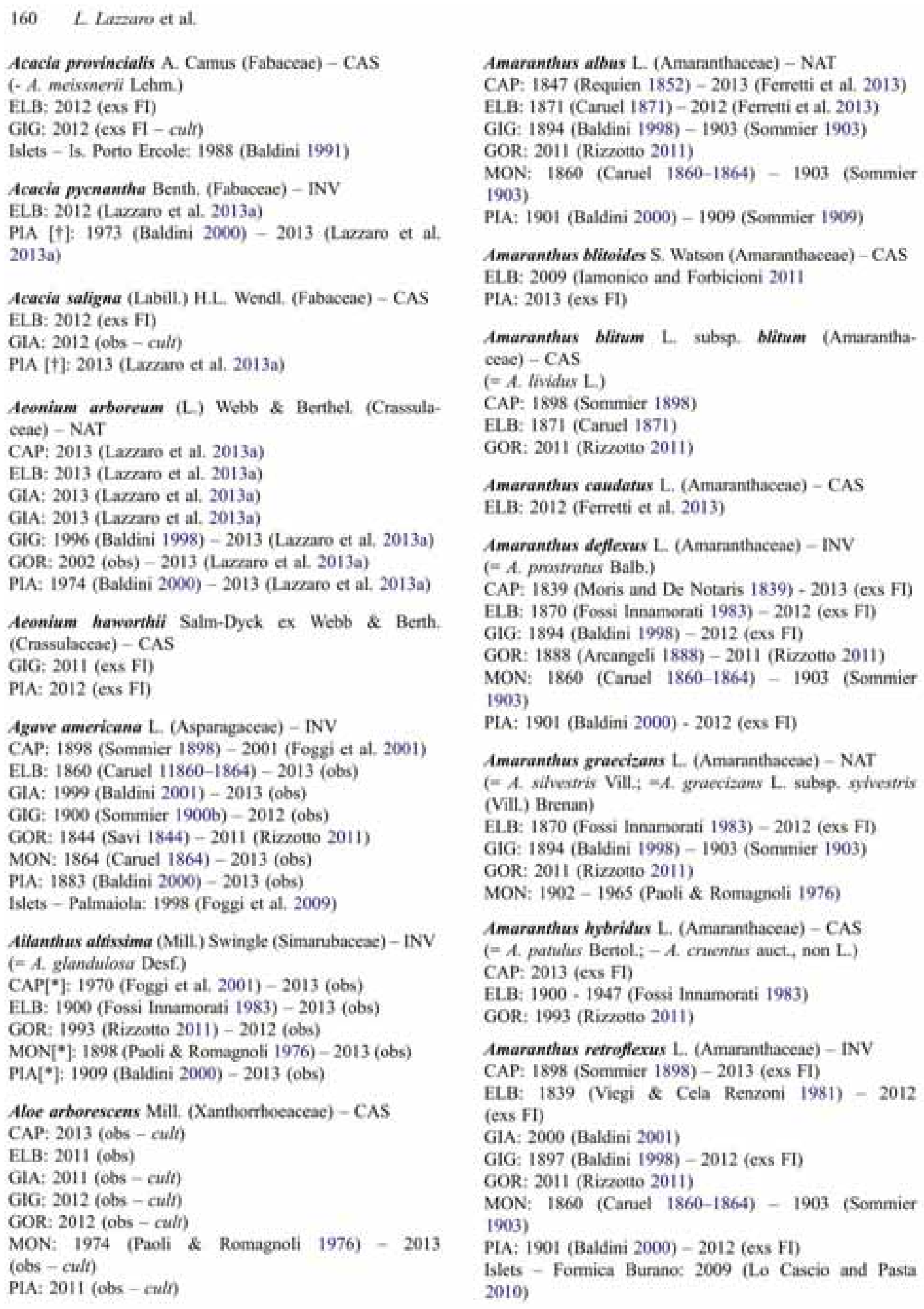
A checklist of the alien flora of the Tuscan Archipelago is presented hereafter as a list of taxa ordered alphabetically and with the family name indicated in parentheses. The invasion status for the whole Tuscan Archipelago is given considering the highest score within the single islands. For each island in which the species is present, the years of the first and of the most recent records are given together with literature reference or information source, i.e. 'obs', referring to recent field observation and 'exs' referring to specimens deposited in the herbarium of Florence (FI) or Wien University (WU). The islands are indicated by the following abbreviations: CAP = Capraia; ELB = Elba; GIA = Giannutri; GIG = Giglio; GOR = Gorgona; MON = Montecristo; PIA = Pianosa, whereas the complete name is given for the islets. The following notations are used when necessary: [†] the species has been recently eradicated from the island; [*] the species is the target of an eradication programme on the island.

In Appendix 1 more information on each species is provided: invasion status for each island and for the whole Archipelago, species life-form, native range and introduction pathway.

Abutilon theophrasti Medik. (Malvaceae) – CAS
GOR: 2011 (Rizzotto 2011)

Acacia dealbata Link (Fabaceae) – INV
ELB: 1998 (Hofmann et al. 1998) – 2012 (Lazzaro et al. 2013a)
GIG: 2013 (Lazzaro et al. 2013a)
Islets – Is. Porto Ercole: 1988 (Baldini 1991)

Acacia melanoxylon R. Br. (Fabaceae) – NAT
CAP: 2011 (Lazzaro et al. 2013a)
ELB: 2012 (Lazzaro et al. 2013a)



Acacia provincialis A. Camus (Fabaceae) – CAS
(- *A. meissnerii* Lehm.)
ELB: 2012 (exs FI)
GIG: 2012 (exs FI – *cult*)
Islets – Is. Porto Ercole: 1988 (Baldini 1991)

Acacia pycnantha Benth. (Fabaceae) – INV
ELB: 2012 (Lazzaro et al. 2013a)
PIA [†]: 1973 (Baldini 2000) – 2013 (Lazzaro et al. 2013a)

Acacia saligna (Labill.) H.L. Wendl. (Fabaceae) – CAS
ELB: 2012 (exs FI)
GIA: 2012 (obs – *cult*)
PIA [†]: 2013 (Lazzaro et al. 2013a)

Aeonium arboreum (L.) Webb & Berthel. (Crassulaceae) – NAT
CAP: 2013 (Lazzaro et al. 2013a)
ELB: 2013 (Lazzaro et al. 2013a)
GIA: 2013 (Lazzaro et al. 2013a)
GIA: 2013 (Lazzaro et al. 2013a)
GIG: 1996 (Baldini 1998) – 2013 (Lazzaro et al. 2013a)
GOR: 2002 (obs) – 2013 (Lazzaro et al. 2013a)
PIA: 1974 (Baldini 2000) – 2013 (Lazzaro et al. 2013a)

Aeonium haworthii Salm-Dyck ex Webb & Berth. (Crassulaceae) – CAS
GIG: 2011 (exs FI)
PIA: 2012 (exs FI)

Agave americana L. (Asparagaceae) – INV
CAP: 1898 (Sommier 1898) – 2001 (Foggi et al. 2001)
ELB: 1860 (Caruel 11860–1864) – 2013 (obs)
GIA: 1999 (Baldini 2001) – 2013 (obs)
GIG: 1900 (Sommier 1900b) – 2012 (obs)
GOR: 1844 (Savi 1844) – 2011 (Rizzotto 2011)
MON: 1864 (Caruel 1864) – 2013 (obs)
PIA: 1883 (Baldini 2000) – 2013 (obs)
Islets – Palmaiola: 1998 (Foggi et al. 2009)

Ailanthus altissima (Mill.) Swingle (Simarubaceae) – INV
(= *A. glandulosa* Desf.)
CAP[*]: 1970 (Foggi et al. 2001) – 2013 (obs)
ELB: 1900 (Fossi Innamorati 1983) – 2013 (obs)
GOR: 1993 (Rizzotto 2011) – 2012 (obs)
MON[*]: 1898 (Paoli & Romagnoli 1976) – 2013 (obs)
PIA[*]: 1909 (Baldini 2000) – 2013 (obs)

Aloe arborescens Mill. (Xanthorrhoeaceae) – CAS
CAP: 2013 (obs – *cult*)
ELB: 2011 (obs)
GIA: 2011 (obs – *cult*)
GIG: 2012 (obs – *cult*)
GOR: 2012 (obs – *cult*)
MON: 1974 (Paoli & Romagnoli 1976) – 2013 (obs – *cult*)
PIA: 2011 (obs – *cult*)

Amaranthus albus L. (Amaranthaceae) – NAT
CAP: 1847 (Requien 1852) – 2013 (Ferretti et al. 2013)
ELB: 1871 (Caruel 1871) – 2012 (Ferretti et al. 2013)
GIG: 1894 (Baldini 1998) – 1903 (Sommier 1903)
GOR: 2011 (Rizzotto 2011)
MON: 1860 (Caruel 1860–1864) – 1903 (Sommier 1903)
PIA: 1901 (Baldini 2000) – 1909 (Sommier 1909)

Amaranthus blitoides S. Watson (Amaranthaceae) – CAS
ELB: 2009 (Iamonico and Forbicioni 2011
PIA: 2013 (exs FI)

Amaranthus blitum L. subsp. ***blitum*** (Amaranthaceae) – CAS
(= *A. lividus* L.)
CAP: 1898 (Sommier 1898)
ELB: 1871 (Caruel 1871)
GOR: 2011 (Rizzotto 2011)

Amaranthus caudatus L. (Amaranthaceae) – CAS
ELB: 2012 (Ferretti et al. 2013)

Amaranthus deflexus L. (Amaranthaceae) – INV
(= *A. prostratus* Balb.)
CAP: 1839 (Moris and De Notaris 1839) - 2013 (exs FI)
ELB: 1870 (Fossi Innamorati 1983) – 2012 (exs FI)
GIG: 1894 (Baldini 1998) – 2012 (exs FI)
GOR: 1888 (Arcangeli 1888) – 2011 (Rizzotto 2011)
MON: 1860 (Caruel 1860–1864) – 1903 (Sommier 1903)
PIA: 1901 (Baldini 2000) - 2012 (exs FI)

Amaranthus graecizans L. (Amaranthaceae) – NAT
(= *A. silvestris* Vill.; =*A. graecizans* L. subsp. *sylvestris* (Vill.) Brenan)
ELB: 1870 (Fossi Innamorati 1983) – 2012 (exs FI)
GIG: 1894 (Baldini 1998) – 1903 (Sommier 1903)
GOR: 2011 (Rizzotto 2011)
MON: 1902 – 1965 (Paoli & Romagnoli 1976)

Amaranthus hybridus L. (Amaranthaceae) – CAS
(= *A. patulus* Bertol.; – *A. cruentus* auct., non L.)
CAP: 2013 (exs FI)
ELB: 1900 - 1947 (Fossi Innamorati 1983)
GOR: 1993 (Rizzotto 2011)

Amaranthus retroflexus L. (Amaranthaceae) – INV
CAP: 1898 (Sommier 1898) – 2013 (exs FI)
ELB: 1839 (Viegi & Cela Renzoni 1981) – 2012 (exs FI)
GIA: 2000 (Baldini 2001)
GIG: 1897 (Baldini 1998) – 2012 (exs FI)
GOR: 2011 (Rizzotto 2011)
MON: 1860 (Caruel 1860–1864) – 1903 (Sommier 1903)
PIA: 1901 (Baldini 2000) – 2012 (exs FI)
Islets – Formica Burano: 2009 (Lo Cascio and Pasta 2010)

Anredera cordifolia (Ten.) Steenis (Basellaceae) – INV
(= *Boussingaultia cordifolia* Ten.)
CAP: 1974 (Pedullà & Garbari 2002) – 2010 (obs)
ELB: 2012 (Lazzaro et al. 2013a)
GIG: 1996 (Baldini 1998)
GOR: 2011 (Rizzotto 2011)
PIA: 2012 (Lazzaro et al. 2013a)

Antirrhinum majus L. subsp. ***majus*** (Plantaginaceae) - NAT
ELB: 1903 (Sommier 1903) – 2012 (Ferretti et al. 2013)
GIG: 2013 (Ferretti et al. 2013)
PIA: 1901 (Baldini 2000) – 1998 (Baldini 2000)

Artemisia verlotiorum Lamotte (Asteraceae) – NAT
ELB: 1981 (Fossi Innamorati 1989)

Arundo donax L. (Poaceae) – INV
(= *Phragmites australis* (Cav.) Steud.)
CAP: 1839 (Moris & De Notaris 1839) – 2013 (obs)
ELB: 1871 (Caruel 1871) – 2013 (obs)
GIA: 1897 (Sommier 1897) – 2000 (Baldini 2001)
GIG: 1894 (Baldini 1998) – 2012 (obs)
GOR: 1844 (Savi 1844) – 2011 (Rizzotto 2011)
MON: 1864 (Caruel 1864) – 1902 (Paoli & Romagnoli 1976)
PIA: 1901 (Sommier 1901) – 2012 (obs)
Islets – Formica Grande: 1898 (Sommier 1903) -1989 (Baldini 1990)

Asclepias fruticosa L. (Apocynaceae) – NAT
(= *Gomphocarpus fruticosus* (L.) W.T. Aiton)
ELB: 2012 (Lazzaro et al. 2013a)
GIG: 1997 (Baldini 1998)

Asparagus falcatus L. (Asparagaceae) – CAS
ELB: 2010 (Forbicioni & Frangini forthcoming)

Austrocylindropuntia subulata (Muehlenpf.) Backeb. (Cactaceae) – NAT
CAP: 2013 (Lazzaro et al. 2013a)
ELB: 2002 (Guiggi 2008) – 2013 (Lazzaro et al. 2013a)
GIA: 2013 (Lazzaro et al. 2013a)
GIG: 2006 (Guiggi 2008) – 2013 (Lazzaro et al. 2013a)
MON: 2013 (Lazzaro et al. 2013a)
PIA: 2013 (Lazzaro et al. 2013a)

Avena byzantina K. Koch (Poaceae) – CAS
GOR: 2011 (Rizzotto 2011)

Avena sativa L. s.l. (Poaceae) – CAS
GIG: 1900 (Sommier 1900b) – 1996 (Baldini 1998)
GOR: 2011 (Rizzotto 2011 – *cult*)
PIA: 1910 (Sommier 1910) – 1998 (Baldini 2000)

Bidens pilosus L. (Asteraceae) – NAT
ELB: 2011 (Pierini & Peruzzi 2012)

Bidens pilosus L. (Asteraceae) – NAT
ELB: 2011 (Pierini and Peruzzi 2012)

Brassica napus L. subsp. ***napus*** (Brassicaceae) – CAS
ELB: 1867 (Fossi Inamorati 1983)
GOR: 1888 (Arcangeli 1888) – 2011 (Rizzotto 2011)
PIA: 1893 (Bolzon 1893b) – 1999 (Baldini 2000)

Brassica oleracea L. (Brassicaceae) – CAS
CAP: 1839 (Moris and De Notaris 1839) – 1970 (Foggi et al. 2001)
ELB: 1860 (Caruel 1860–1864)
GOR: 1844 (Savi 1844) – 2011 (Rizzotto 2011)

Broussonetia papyrifera (L.) Vent. (Moraceae) – NAT
CAP: 1898 (Sommier 1898) – 2001 (Foggi et al. 2001)
PIA: 2012 (Ferretti et al. 2013)

Callitropsis glabra (Sudw.) D.P. Little (Cupressaceae) – NAT
ELB: 2012 (Lazzaro et al. 2013a)
GIA: 2012 (obs – *cult*)
GIG: 2012 (obs – *cult*)

Campsis radicans (L.) Bureau (Bignoniaceae) – NAT
GIA: 2012 (obs – *cult*)
GIG: 2011 (Lazzaro et al. 2013a)
MON: 2013 (obs)

Canna indica L. (Cannaceae) – CAS
ELB: 2012 (Ferretti et al. 2013)
GIA: 2012 (obs – *cult*)

Carpobrotus acinaciformis (L.) L. Bolus (Aizoaceae) – INV
CAP[*]: 2001 (Foggi et al. 2001) – 2013 (Lazzaro et al. 2013a)
ELB: 1882 (Fossi Innamorati 1983) – 2013 (Lazzaro et al. 2013a)
GIA: 2000 (Baldini 2001) – 2013 (Lazzaro et al. 2013a)
GIG: 1894 (Sommier 1894b) – 2013 (Lazzaro et al. 2013a)
GOR: 2013 (Lazzaro et al. 2013a)
MON: 1964 (Paoli & Romagnoli 1976) – 2013 (Lazzaro et al. 2013a)
PIA [†]: 1883 (Baldini 2000) – 2013 (Lazzaro et al. 2013a)
Islets – Palmaiola – Cerboli: 1998 (Foggi et al. 2009)

Carpobrotus edulis (L.) N.E. Br. (Aizoaceae) – INV
ELB: 2010 (obs)
GIA: 2013 (Lazzaro et al. 2013a)
GIG: 2013 (Lazzaro et al. 2013a)

Casuarina equisetifolia L. (Casuarinaceae) – NAT
PIA: 1999 (Baldini 2000) – 2012 (obs)

Catharanthus roseus (L.) G. Don (Apocynaceae) – NAT
ELB: 2012 (Ferretti et al. 2013)

Cenchrus longisetus M.C. Johnst. (Poaceae) – CAS
(= *Pennisetum villosum* Fresen.)
ELB: 2012 (Forbicioni et al. forthcoming)

Chamaesyce humifusa (Willd. ex Schltr.) Prokh. (Euphorbiaceae) – NAT
ELB: 2012 (Ferretti et al. 2013)

Chamaesyce maculata (L.) Small (Euphorbiaceae) – INV
(-*Euphorbia thymifolia* auct., non Burm.)
CAP: 1898 (Sommier 1898) – 2013 (obs)
ELB: 2012 (Ferretti et al. 2013)
PIA: 2012 (Ferretti et al. 2013)

Chamaesyce prostrata (Aiton) Small (Euphorbiaceae) – NAT
GIA: 2000 (Baldini 2001)
GIG: 2012 (Ferretti et al. 2013)

Chasmanthe aethiopica (L.) N.E. Br. (Iridaceae) – NAT
(≡ *Antholyza aethiopica* L.)
CAP: 1898 (Béguinot 1901) – 2012 (Lastrucci et al. 2012)
ELB: 1900 (Sommier 1900a) – 2012 (obs)

Citrullus lanatus (Thunb.) Matsum. & Nakai (Cucurbitaceae) – CAS
(= *Citrullus vulgaris* Schrad.; – *Cucumis anguria* auct., non L.)
ELB: 1808 (Thiébaut De Berneaud 1808) – 2012 (Ferretti et al. 2013)
GIA: 2012 (Ferretti et al. 2013)
GOR: 1888 (Arcangeli 1888) – 2012 (Ferretti et al. 2013)

Cortaderia selloana (Schult. & Schult. f.) Asch. & Graebn. (Poaceae) – INV
CAP: 2010 (obs – *cult*)
ELB: 2012 (Lazzaro et al. 2013a)
GOR: 2012 (obs – *cult*)

Cotoneaster coriaceus Franch. (Rosaceae) – NAT
ELB: 2008 (Galasso et al. 2011)

Cotyledon orbiculata L. (Crassulaceae) – CAS
CAP: 2013 (Lazzaro et al. 2013a)
ELB: 2013 (Lazzaro et al. 2013a)
GIA: 2013 (Lazzaro et al. 2013a)
GIG: 2013 (Lazzaro et al. 2013a)
PIA: 2011 (Lazzaro et al. 2013a)

Crassula lycopodioides Lam. (Crassulaceae) – CAS
ELB: 2012 (Lazzaro et al. 2013a)
GIA: 2012 (Lazzaro et al. 2013a)
GIG: 2013 (Lazzaro et al. 2013a)

Crataegus azarolus L. (Rosaceae) – NAT
CAP: 1898 (Sommier 1898) – 2013 (obs)

Cucurbita maxima Duchesne (Cucurbitaceae) – CAS
ELB: 2012 (Ferretti et al. 2013)
GOR: 2012 (Ferretti et al. 2013)

Cupressus sempervirens L. (Cupressaceae) – NAT
CAP: 1898 (Sommier 1898) – 2013 (Lazzaro et al. 2013a)
ELB: 1962 (Fossi Innamorati 1983) – 2013 (Lazzaro et al. 2013a)
GIA: 2000 (Baldini 2001; *cult*)
GIG: 1996 (Baldini 1998) – 2013 (Lazzaro et al. 2013a)
GOR: 1888 (Arcangeli 1888) – 2013 (Lazzaro et al. 2013a)
MON: 1957 (Fabbri 1963) – 2013 (obs)
PIA: 2012 (Lazzaro et al. 2013a)

Cuscuta epilinum Weihe (Convolvulaceae) – CAS
CAP: 1989 (Mannocci & Barsotti 1989)
ELB: 1863 (Fossi Innamorati 1989)
GOR: 1902 (Sommier 1902)

Cyperus involucratus Rottb. (Cyperaceae) – NAT
ELB: 2013 (Ferretti et al. 2013)

Datura ferox L. (Solanaceae) – NAT
PIA: 2012 (Ferretti et al. 2013)

Datura stramonium L. subsp. ***stramonium*** (Solanaceae) – NAT
CAP: 1999 (Foggi et al. 2001) – 2013 (obs)
ELB: 1870 (Fossi Innamorati 1989)
GIG: 1894 (Baldini 1998)
GOR: 1993 (Rizzotto 2011)
MON: 1860 (Caruel 1860–1864) – 1903 (Sommier 1903)
PIA: 2012 (Ferretti et al. 2013)

Datura wrightii Regel (Solanaceae) – NAT
CAP: 2013 (obs)
GIA: 2011 (Lazzaro et al. 2013a)
GIG: 2013 (Lazzaro et al. 2013a – *cult*)

Delairea odorata Lem. (Asteraceae) – NAT
(= *Senecio mikanioides* Walp.)
ELB: 2010 (Frangini and Carta 2011)

Delosperma cooperi (Hook. f.) L. Bolus (Aizoaceae) – CAS
CAP: 2001 (Foggi et al. 2001)
GIG: 2012 (obs – *cult*)
PIA: 2011 (obs – *cult*)

Drosanthemum floribundum (Haw.) Schwantes (Aizoaceae) – NAT
ELB: 2012 (Lazzaro et al. 2013a)
GIG: 2011 (Lazzaro et al. 2013a)

Dysphania ambrosioides (L.) Mosyakin & Clemants (Amaranthaceae) – INV
CAP: 2013 (obs)
ELB: 2008 (Lazzaro et al. 2013a)

Dysphania multifida (L.) Mosyakin & Clemants (Amaranthaceae) – NAT
(= *Chenopodium multifidum* L.; – *Chenopodium botrys* auct., non L.)
GIG: 1900 (Sommier 1900b) – 1996 (Baldini 1998)

Eleusine indica (L.) Gaertn. subsp. ***indica*** (Poaceae) – NAT
PIA: 2012 (Ferretti et al. 2013)

Erigeron bonariensis L. (Asteraceae) – INV
(= *E. linifolius* Willd.; ≡ *Conyza bonariensis* (L.) Cronquist; = *C. ambigua* DC.)
CAP: 1839 (Moris & De Notaris 1839) – 2013 (exs FI)
ELB: 1900 (Fossi Innamorati 1989) – 2013 (exs FI)
GIA: 1886 (Baldini 2001) – 2012 (obs)
GIG: 1894 (Baldini 1998) – 2011 (obs)
GOR: 1888 (Arcangeli 1888) – 2011 (Rizzotto 2011)
MON: 1864 (Caruel 1864) – 2013 (exs FI)
PIA: 1901 (Sommier 1901) – 1999 (Baldini 2000)
Islets – Formica Burano: 2009 (Lo Cascio and Pasta 2010)

Erigeron canadensis L. (Asteraceae) – NAT
(≡ *Conyza canadensis* (L.) Cronquist)
CAP: 1839 (Moris & De Notaris 1839) – 2001 (Foggi et al. 2001)
ELB: 1893 (Bolzon 1893a)
GIA: 1882 (Baldini 2001)
GIG: 1894 (Baldini 1998)
GOR: 1888 (Arcangeli 1888)
MON: 1864 (Caruel 1864)
PIA: 1901 (Sommier 1901)

Erigeron karvinskianus DC. (Asteraceae) – INV
ELB: 1986 (Fossi Innamorati 1989) – 2013 (exs FI)

Erigeron sumatrensis Retz. (Asteraceae) – INV
CAP: 2013 (obs)
ELB: 1972 – 2012 (Ferretti et al. 2013)
GIA: 2013 (Ferretti et al. 2013)
GIG: 2012 (Ferretti et al. 2013)
GOR: 2012 (Ferretti et al. 2013)
MON: 2013 (Ferretti et al. 2013)
PIA: 2012 (Ferretti et al. 2013)

Erysimum cheiri (L.) Crantz (Brassicaceae) – NAT
(≡ *Cheiranthus cheiri* L.)
CAP: 1898 (Sommier 1898) – 2001 (Foggi et al. 2001)
ELB: 1871 (Caruel 1871) – 1972 (Fossi Innamorati 1983)
GIG: 1900 (Sommier 1900b) – 1997 (Baldini 1998)
GOR: 1844 (Savi 1844) – 2011 (Rizzotto 2011)
PIA: 1892 (Bolzon 1892) – 1998 (Baldini 2000)

Erythrostemon gilliesii (Wall. ex Hook.) Klotzsch (Fabaceae) – CAS
ELB: 2012 (Ferretti et al. 2013)

Eucalyptus camaldulensis Dehnh. (Myrtaceae) – NAT
CAP: 2013 (obs)
ELB: 1964 (Fossi Innamorati 1983) – 2013 (obs)
GIA: 2013 (exs FI – *cult*)
GIG: 1996 (Baldini 1998) – 2012 (obs)
PIA: 1999 (Baldini 2000) – 2012 (obs)

Eucalyptus globulus Labill. (Myrtaceae) – NAT
CAP: 1989 (Foggi et al. 2001) – 2013 (obs – *cult*)
ELB: 1891 (Fossi Innamorati 1983) – 2013 (exs FI – *cult*)
GIG: 1996 (Baldini 1998) – 2012 (obs – *cult*)
MON: 1957 (Fabbri 1963) – 2013 (exs FI)

Fallopia baldschuanica (Regel) Holub (Polygonaceae) – NAT
ELB: 2012 (Ferretti et al. 2013)

Freesia alba (G.L. Mey.) Gumbl. (Iridaceae) – CAS
CAP: 2012 (Peruzzi & Gestri 2013)

Gazania linearis (Thumb.) Druce (Asteraceae) – CAS
ELB: 2012 (Peruzzi et al. 2013)

Gleditsia triacanthos L. (Fabaceae) – CAS
ELB: 2012 (Ferretti et al. 2013)

Helianthus annuus L. (Asteraceae) – CAS
PIA: 1998 (Baldini 2000)

Helianthus tuberosus L. (Asteraceae) – NAT
ELB: 2012 (Ferretti et al. 2013)
GOR: 1993 (Rizzotto 2011)

Ipomoea indica (Burm.) Merr. (Convolvulaceae) – NAT
ELB: 2008 (Lazzaro et al. 2013a)

Iris germanica L. (Iridaceae) – NAT
CAP: 1839 (Moris & De Notaris 1839) – 2012 (Peruzzi and Gestri 2013)
ELB: 1882 (Fossi Innamorati 1991) – 2013 (obs)
GIA: 1897 (Sommier 1897) – 2013 (obs)
GIG: 1900 (Sommier 1900b) – 2013 (obs)
GOR: 1993 (Rizzotto 2011)
MON: 2012 (obs)
PIA: 1892 (Bolzon 1892) – 2013 (obs)

Kalanchoē × houghtonii D.B. Ward (Crassulaceae) – NAT
ELB: 2008 (Lazzaro et al. 2013a)
GIA: 2011 (Lazzaro et al. 2013a)
PIA: 2012 (Lazzaro et al. 2013a)

Lantana camara L. (Verbenaceae) – NAT
CAP: 2013 (obs – *cult*)
GIA: 2013 (Lazzaro et al. 2013a)
GIG: 2012 (Lazzaro et al. 2013a)

Lepidium didymum L. (Brassicaceae) – NAT
ELB: 2012 (Ferretti & Foggi 2013)
MON: 2011 (Carta et al. 2012)

Ligustrum lucidum W.T. Aiton (Oleaceae) – NAT
ELB: 2008 (Lazzaro et al. 2013a)

Lonicera japonica Thunb. (Caprifoliaceae) – NAT
ELB: 2012 (Ferretti & Foggi 2013)

Lupinus albus L. subsp. ***albus*** (Fabaceae) – CAS
ELB: 1893 (Bolzon 1893a)
GIG: 1900 (Sommier 1900b)
PIA: 1902 (Baldini 2000) – 1998 (Baldini 2000)

Melia azedarach L. (Meliaceae) – CAS
CAP: 2013 (obs FI)

Mesembryanthemum cordifolium L. f. (Aizoaceae) – INV
ELB: 2008 (Lazzaro et al. 2013a)
GIA: 2013 (Lazzaro et al. 2013a)
GIG: 2013 (Lazzaro et al. 2013a)
GOR: 1990 (Moggi et al. 1991) – 2013 (Lazzaro et al. 2013a)
PIA: 2013 (Lazzaro et al. 2013a)

Mirabilis jalapa L. (Nyctaginaceae) – INV
CAP: 2013 (obs)
ELB: 2008 (Lazzaro et al. 2013a)
GIG: 2013 (Lazzaro et al. 2013a)
GOR: 1993 (Rizzotto 2011) – 2013 (Lazzaro et al. 2013a)
PIA: 2013 (Lazzaro et al. 2013a)

Morus alba L. (Moraceae) – CAS
CAP: 1970 – 2001 (Foggi et al. 2001)
ELB: 1900 – 1974 (Fossi Innamorati 1983)
GOR: 1993 (Rizzotto 2011 – *cult*)
MON: 1902 (Paoli & Romagnoli 1976) – 2013 (obs – *cult*)
PIA: 1836 (Sommier 1909)

Myoporum insulare R. Br. (Scrophulariaceae) – CAS
ELB: 2012 (Lazzaro et al. 2013a)
GIA: 2011 (Lazzaro et al. 2013a)

Nicotiana glauca Graham (Solanaceae) – INV
CAP: 1976 (Montelucci 1976) – 2013 (exs FI)

Nothoscordum gracile (Aiton) Stearn (Amaryllidaceae) – CAS
PIA: 2012 (Ferretti et al. 2013)

Opuntia dillenii (Ker-Gawl.) Haw. (Cactaceae) – NAT
GIG: 2003 (Guiggi 2008) – 2012 (obs)

Opuntia ficus-indica (L.) Mill. (Cactaceae) – INV
(- *Opuntia inermis* auct., non (DC.) DC.; – *O. vulgaris* auct., non Mill.)
CAP: 1898 (Sommier 1898) – 2013 (obs)
ELB: 1891 (Fossi Innamorati 1983) – 2013 (obs)
GIA: 1999 (Baldini 2001) – 2013 (obs)
GIG: 1894 (Sommier 1894a) – 2013 (obs)
GOR: 1888 (Arcangeli 1888) – 2011 (Rizzotto 2011)
MON [**]: 1864 (Caruel 1864) – 1974 (Paoli & Romagnoli 1976)
PIA: 1901 (Sommier 1901) – 2012 (obs)
Islets – Palmaiola: 1998 (Foggi et al. 2009)
[**]: Although the species is reported in 1974, nowadays it appears to be absent from the island. The cited report is probably referred to O. monacantha.

Opuntia leucotricha DC. (Cactaceae) – NAT
ELB: 2013 (Lazzaro et al. 2013b)

Opuntia monacantha (Willd.) Haw. (Cactaceae) – NAT
(- *O. maxima* auct., non Mill.)
CAP: s.d. (Guiggi 2008 – *cult*) – 2013 (obs – *cult*)
ELB: 1849 (Guiggi 2008) – 2013 (Lazzaro et al. 2013a)
GIA: 2013 (Lazzaro et al. 2013a)
MON: 2013 (Lazzaro et al. 2013a)
PIA: 2013 (Lazzaro et al. 2013a)

Opuntia phaeacantha Engelm. (Cactaceae) – INV
GIA: 2007 (Guiggi 2010)

Opuntia stricta (Haw.) Haw. (Cactaceae) – INV
(= *O. inermis* (DC.) DC.; – *O. vulgaris* auct., non Mill.)
CAP: 1839 (Moris and De Notaris 1839) – 2013 (Lazzaro et al. 2013a)
GIG: 2013 (Lazzaro et al. 2013a)
PIA: 1901 (Guiggi 2008)

Osteospermum ecklonis (DC.) Norl. (Asteraceae) – CAS
CAP: 2013 (obs)
ELB: 2012 (obs)
GIA: 2012 (obs – *cult*)
GIG: 2012 (obs – *cult*)

Oxalis articulata Savigny (Oxalidaceae) – NAT
CAP: 2001 (Foggi et al. 2001)
ELB: 2008 (Lazzaro et al. 2013a)
GIG: 1996 (Baldini 1998) – 2013 (Lazzaro et al. 2013a)
GOR: 1993 (Rizzotto 2011) – 2013 (Lazzaro et al. 2013a)
MON: 1965 – 2012 (Lazzaro et al. 2013a)
PIA: 2013 (Lazzaro et al. 2013a)

Oxalis bowiei Herb. ex Lindl. (Oxalidaceae) – NAT
ELB: 2012 (Ferretti et al. 2013)

Oxalis debilis Kunth (Oxalidaceae) – NAT
ELB: 2008 (Lazzaro et al. 2013a)

Oxalis dillenii Jacq. (Oxalidaceae) – NAT
ELB: 2008 (Lazzaro et al. 2013a)
GIA: 2000 (Baldini 2001) – 2012 (obs)

Oxalis latifolia Kunth (Oxalidaceae) – NAT
ELB: 2013 (Lazzaro et al. 2013a)
GIG: 2013 (Lazzaro et al. 2013a)

Oxalis pes-caprae L. (Oxalidaceae) – INV
CAP: 1995 (exs FI) – 2012 (Peruzzi and Gestri 2013)
ELB: 1955 (exs WU) – 2012 (obs)
GIA: 1999 (Baldini 2001) – 2012 (obs)
GIG: 1996 (Baldini 1998) – 2012 (obs)
GOR: 1987 (exs FI) – 2011 (Rizzotto 2011)
MON: 1957 (Fabbri 1963) – 2013 (obs)
PIA: 1974 (Sabato 1977) – 2012 (obs)

Paraserianthes lophantha (Willd.) I.C. Nielsen (Fabaceae) – NAT
CAP: 2011 (Lazzaro et al. 2013a)
ELB: 2009 (Frangini et al. 2010)

Parthenocissus quinquefolia (L.) Planch. (Vitaceae) – NAT
ELB: 2012 (Lazzaro et al. 2013a)
GIA: 2013 (Lazzaro et al. 2013a)
GIG: 2013 (Lazzaro et al. 2013a)
PIA: 2012 (Lazzaro et al. 2013a)

Paspalum dilatatum Poir. (Poaceae) – NAT
ELB: 2012 (Ferretti et al. 2013)
GIG: 2012 (Ferretti et al. 2013)

Paspalum distichum L. (Poaceae) – NAT
CAP: 1976 (Montelucci 1976) – 2013 (obs)
ELB: 1947 (Fossi Innamorati 1991) – 2013 (Ferretti et al. 2013)
GIG: 1996 (Baldini 1998)
GOR: 2012 (Ferretti et al. 2013)

Passiflora caerulea L. (Passifloraceae) – NAT
CAP: 2013 (obs – *cult*)
ELB: 2012 (Lazzaro et al. 2013a)

Pelargonium zonale (L.) L'Hér. (Geraniaceae) – CAS
CAP: 2013 (obs)
ELB: 2012 (obs)
GIA: 2013 (obs)
GIG: 2011 (obs)
PIA: 2012 (obs)

Phacelia tanacetifolia Benth. (Boraginaceae) – CAS
GOR: 2011 (Rizzotto 2011)

Phoenix canariensis Chabaud (Arecaceae) – CAS
ELB: 1981 (Fossi Innamorati 1991) – 2012 (obs)
GIA: 2012 (obs – *cult*)
GIG: 2012 (obs – *cult*)

Phytolacca americana L. (Phytolaccaceae) – INV
ELB: 1870 (Fossi Innamorati 1983) – 2012 (obs)
GIG: 1900 (Sommier 1900b) – 1996 (Baldini 1998)
GOR: 1993 (Rizzotto 2011)
MON: 1964 (Paoli & Romagnoli 1976) – 2013 (obs)

Pinus canariensis C. Sm. (Pinaceae) – NAT
ELB: 1950 (Gatteschi & Arretini 1990) – 2012 (exs FI)
GIG: 2012 (obs – *cult*)

Pisum sativum L. subsp. ***sativum*** (Fabaceae) – CAS
(= *P. arvense* L.)
CAP: 1839 (Moris and De Notaris 1839)
ELB: 2012 (obs – *cult*)
GIG: 1897 – 1996 (Baldini 1998)
PIA: 1901 (Sommier 1901) – 1998 (Baldini 2000)

Pittosporum tobira (Thunb.) W.T. Aiton (Pittosporaceae) – NAT
CAP: 2013 (Lazzaro et al. 2013a)
ELB: 2013 (Lazzaro et al. 2013a)
GIA: 2013 (Lazzaro et al. 2013a)
GIG: 2013 (Lazzaro et al. 2013a)

Prunus armeniaca L. (Rosaceae) – CAS
CAP: 2012 (obs – *cult*)
ELB: 1898 (Fossi Innamorati 1983) – 2012 (obs)
GIG: 1900 (Sommier 1900b) – 2012 (obs – *cult*)
MON: 2012 (obs – *cult*)
PIA: 2013 (obs)

Prunus cerasus L. (Rosaceae) – CAS
CAP: 1896 (Viegi & Cela Renzoni 1981) – 2001 (Foggi et al. 2001)
ELB: 1898 (Fossi Innamorati 1983) – 2011 (obs)
GIG: 1894 – 1996 (Baldini 1998)
PIA: 1909 (Sommier 1909) – 1998 (Baldini 2000)

Prunus domestica L. s.l. (Rosaceae) – NAT
CAP: 1839 (Moris & De Notaris 1839) – 2013 (obs)
ELB: 1900 (Fossi Innamorati 1983)
GIG: 1894 (Viegi & Cela Renzoni 1981) – 2012 (obs)

Robinia pseudoacacia L. (Fabaceae) – INV
CAP: 1970 (Foggi et al. 2001) – 2001 (Foggi et al. 2001)
ELB: 1903 (Sommier 1903) – 2013 (obs)
GIG: 1996 (Baldini 1998) – 2012 (obs)
GOR: 1899 (Sommier 1899) – 2011 (Rizzotto 2011)
MON: 2012 (obs – *cult*)
PIA: 1909 (Sommier 1909) – 2013 (obs)

Senecio angulatus L. f. (Asteraceae) – INV
(- *S. mikanioides* auct., non Walp.)
CAP: 1970 (Foggi et al. 2001) – 2013 (Ferretti et al. 2013)
ELB: 1971 (Fossi Innamorati 1989) – 2013 (Ferretti et al. 2013)
GIA: 1999 (Baldini 2001) – 2013 (Ferretti et al. 2013)
GIG: 1996 (Baldini 1998) – 2013 (Lazzaro et al. 2013a)
GOR: 2012 (Ferretti et al. 2013)
PIA: 1998 (Baldini 2000) – 2013 (Ferretti et al. 2013)

Setaria parviflora (Poir.) Kerguélen (Poaceae) – NAT
ELB: 2012 (Ferretti et al. 2013)

Solanum lycopersicum L. (Solanaceae) – CAS
CAP: 1839 (Moris & De Notaris 1839) – 2013 (obs)
ELB: 2013 (Ferretti et al. 2013)
GIA: 2013 (Ferretti et al. 2013)
GIG: 1900 (Sommier 1900b) – 2013 (Ferretti et al. 2013)
GOR: 1844 (Savi 1844) – 2013 (Ferretti et al. 2013)
MON: 2011 (obs – *cult*)
PIA: 2013 (obs)

Sorghum bicolor (L.) Moench (Poaceae) – CAS
CAP: 1976 (Montelucci 1976 – *cult*)
GOR: 2011 (Rizzotto 2011)

Sorghum halepense (L.) Pers. (Poaceae) – INV
CAP: 1839 (Moris & De Notaris 1839) – 2013 (Ferretti et al. 2013)

ELB: 1839 (Fossi Innamorati 1991) – 2013 (obs)
GIA: 2000 (Baldini 2001)
MON: 1860 (Caruel 1860–1864) – 1903 (Sommier 1903)
PIA: 1901 (Sommier 1901) – 2013 (Ferretti et al. 2013)

Symphyotrichum squamatum (Spreng.) G.L. Nesom (Asteraceae) – INV
(≡ *Aster squamatus* (Spreng.) Hieron.)
CAP: 2013 (obs)
ELB: 1964 (Fossi Innamorati 1989) – 2012 (obs)
GIA: 2012 (obs)
GOR: 1993 (Rizzotto 2011) – 2012 (obs)
PIA: 1999 (Baldini 2000) – 2013 (obs)

Tamarix parviflora DC. (Tamaricaceae) – CAS
ELB: 1972 (Fossi Innamorati 1983) – 1981 (Fossi Innamorati 1983)
GIG: 1996 – 2009 (Gargano et al. 2009 – *cult*)

Tanacetum parthenium (L.) Sch. Bip. (Asteraceae) – CAS
ELB: 1898 (Fossi Innamorati 1989)
GIG: 1894 – 1996 (Baldini 1998)

Tetragonia tetragonoides (Pallas) Kuntze (Aizoaceae) – CAS
GIA: 2012 (Lazzaro et al. 2013a)

Triticum aestivum L. (Poaceae) – CAS
(= *Triticum vulgare* Vill.)
ELB: 1900 (Fossi Innamorati 1991) – 2012 (obs)
MON: 1965 (Paoli & Romagnoli 1976) – 2011 (exs FI)
PIA: 1999 (Baldini 2000)

Tropaeolum majus L. (Tropaeolaceae) – CAS
CAP: 2012 (Lastrucci et al. 2012)
ELB: 2008 (Lazzaro et al. 2013a)
GIA: 2012 (obs – *cult*)

Vachellia karroo (Hayne) Banfi & Galasso (Fabaceae) – CAS
ELB: 2012 (Lazzaro et al. 2013a)

Veronica persica Poir. (Plantaginaceae) – INV
CAP: 1970 (Foggi et al. 2001) – 1989 (Mannocci and Barsotti 1989)
ELB: 1901 (Fossi Innamorati 1989) – 2012 (obs)
GOR: 1993 (Rizzotto 2011)

Vitis riparia Michx. × ***Vitis rupestris*** Scheele (Vitaceae) – INV
CAP: 2012 (Lastrucci et al. 2012)
ELB: 2012 (obs)
GIA: 2012 (obs – *cult*)
GIG: 2012 (obs)
GOR: 2012 (obs – *cult*)
PIA: 2012 (obs – *cult*)

Xanthium orientale L. subsp. ***italicum*** (Moretti) Greuter (Asteraceae) – NAT
(≡ X. *strumarium* subsp. *italicum* (Moretti) D. Löve)
CAP: 1989 (Mannocci & Barsotti 1989) – 2013 (obs)
ELB: 1870 (Fossi Innamorati 1989) – 2012 (exs FI)
GIG: 1895 – 1996 (Baldini 1998)
PIA: 2013 (exs FI)

Xanthium spinosum L. (Asteraceae) – NAT
ELB: 1900 – 1972 (Fossi Innamorati 1989)
GIG: 1897 (Sommier 1900b)
GOR: 1993 (Rizzotto 2011) – 2012 (exs FI)
MON: 1864 (Caruel 1864) – 1903 (Sommier 1903)

Yucca aloifolia L. (Asparagaceae) – CAS
CAP: 1898 (Sommier 1898 – *cult* – 2013 obs – *cult*)
ELB: 2012 (obs)
GIA: 2012 (obs – *cult*)
GIG: 2012 (obs – *cult*)
MON: 1974 (Paoli & Romagnoli 1976 – 2013 obs – *cult*)

Zantedeschia aethiopica (L.) Spreng. (Araceae) – INV
CAP: 2012 (Lastrucci et al. 2012)
ELB: 2012 (Ferretti et al. 2013)

Ziziphus jujuba Mill. (Rhamnaceae) – NAT
ELB: 1808 (Thiébaut De Berneaud 1808) – 2013 (Lazzaro et al. 2013a)
PIA: 2012 (Lazzaro et al. 2013a)

Taxonomy and life forms

The checklist of the alien flora of the Tuscan Archipelago consists of 141 taxa, including the hybrid *Vitis riparia* Michx. × *Vitis rupestris* Scheele, which includes all the hybrids used for agricultural purposes. According to our estimation of 1400 taxa representing the total spontaneous flora of the Tuscan Archipelago, alien taxa account for 10% of the total flora. The number of alien taxa and their percentage on the total flora for each island and for the entire Archipelago are reported in Figure 2.

Three taxa were gymnosperms (*Callitropsis glabra*, *Cupressus sempervirens* and *Pinus canariensis*), while all the other 138 were angiosperms. We listed 101 genera in 49 families. The genera containing most taxa were *Amaranthus* (eight taxa), *Opuntia* (six taxa), *Oxalis* (six taxa), *Acacia* (five taxa) and *Erigeron* (four taxa) with all the others containing from three to one taxa per genera. The most represented family was the Asteraceae (16 taxa), followed by Poaceae (12 taxa), Fabaceae (12 taxa), Amaranthaceae (10 taxa), Cactaceae (seven taxa), Oxalidaceae and Aizoaceae (six taxa). Nine families contained only two taxa and 26 families were present with only one taxon. Most of the species were trees and shrubs (44 species) and annual herbs (42 species) (Figure 3), and the distributions of the three invasion statuses were similar across the different life forms.

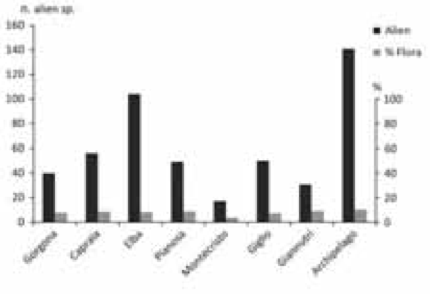


Figure 2. Number of alien taxa, and their percentage of the total spontaneous flora, for each island and for the whole Tuscan Archipelago.

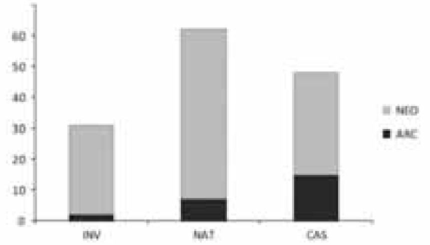


Figure 4. Number of alien taxa of the Tuscan Archipelago according to their invasion status (INV = invasive, NAT = naturalized, CAS = casual), classified according to the residence time (ARC = archaeophytes, NEO = neophytes).

Invasion status

Thirty-one taxa were assessed as invasive in the Archipelago (two archaeophytes and 29 neophytes), 62 as naturalized (seven archaeophytes and 55 neophytes) and 48 as casual (15 archaeophytes and 33 neophytes) (Figure 4). Elba Island and Capraia hosted the highest numbers of invasive taxa (24 and 19 taxa, respectively), while Montecristo hosted only one invasive species, *Ailanthus altissimus*, which is the target of an eradication programme (Table 2). With regard to the residence time, 117 (83%) were neophytes and the remaining 24 (17%) were archaeophytes.

Table 2. Number of taxa for invasion status

	ARC	Cap	Elb	Gia	Gig	Gor	Mon	Pia	Isl
INV	31	19	24	9	13	4	1	13	0
NAT	62	18	46	11	25	19	9	20	5
CAS	48	19	34	12	12	17	7	16	1
TOT	**141**	**56**	**104**	**32**	**50**	**40**	**17**	**49**	**6**
CULT	0	9	2	12	12	5	6	3	0
NR	0	2	9	1	6	2	8	4	0

Notes: (INV = invasive, NAT = naturalized, CAS = casual, CULT = only cultivated and NR = not retrieved in the island after 1950; status as given according to Pyšek et al. 2004a) for the whole Archipelago, the seven main islands and for the islets (ARC = whole Archipelago, Cap = Capraia, Elb = Elba, Gia = Giannutri, Gig = Giglio, Gor = Gorgona, Mon = Montecristo, Pia = Pianosa, Isl = islets).

Native range and introduction pathway

As to the native range (Appendix 1; Figure 5), most of the alien plants of the Tuscan Archipelago (61, representing the 43% of the taxa) came from the Americas, mainly from South America (23) and North America (21). Thirty-one taxa (22%) were native to Africa, mostly to South Africa (18), and 20 (14%) came from Asia, 11 (8%) from the Mediterranean, 11 (8%) from Australia and six (4%) from Eurasia.

Most of the taxa (71%) were voluntarily introduced, mainly as ornamental species (48%), but also as forestry

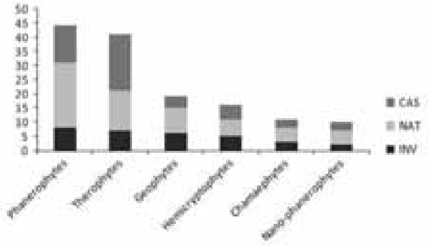


Figure 3. Distribution of the alien taxa of the Tuscan Archipelago according to Raunkiaer's life forms (Raunkiaer 1934), classified according to their invasion status.

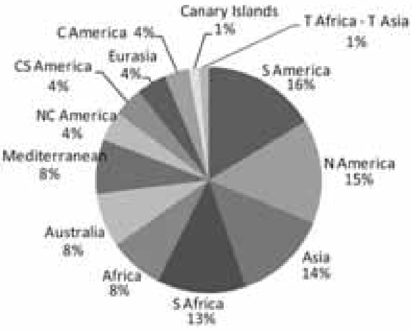


Figure 5. Percentage of the alien taxa of the Tuscan Archipelago according to the native range.

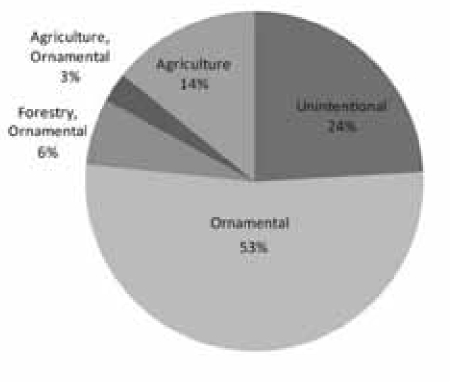


Figure 6. Percentage of the alien taxa of the Tuscan Archipelago according to introduction pathways.

and ornamental species (13%), agriculture (7%) and both agricultural and ornamental purposes (3%). Only 29% of the taxa were introduced unintentionally into the Archipelago (Appendix 1; Figure 6).

Discussion

According to the present inventory, the alien flora of the Tuscan Archipelago consists of 141 alien taxa. This number is particularly high if compared to the 64 species listed in Viegi and Cela Renzoni (1981). Moreover, only 30 of those 64 species were confirmed by our work, whereas 14 have not been retrieved since 1950, and 20 are cryptogenic or only-cultivated species. Therefore we are reporting an increase of 111 species in the last 30 years. However, according to the year of the record, 64 species had already been recorded before 1981; so the actual increase of species should be reduced to 77 species, of which 67 (representing almost half of the species listed herewith) have been reported since 2000.

These data reflect, on the one hand, the increase in the interest in and the inventory efforts on alien species, and, on the other hand, the deep socio-economic changes in the Tuscan Archipelago since the 1960s. In this framework, it is noteworthy that most of the species established in the last decades had been introduced as ornamentals (53 of the 77 species retrieved after 1981) while only six (out a total of 20) had been introduced for agricultural purposes. The number of alien species introduced for agriculture or forestry is proportionally higher in the species introduced before the 1980s, and could be even higher considering the not retrieved species. This evidence is consistent with the general change that involved the Mediterranean islands in the last decades, with a transition from an economy based largely on agricultural exploitation to one based on tourism development (Papayannis & Sorocou 2008), so determining a shift in the land use and in the introduction pathways of alien species.

According to Moody (2000), the number of non-native taxa on islands is related to their size; indeed the largest islands of the Tuscan Archipelago (Elba, Giglio and Capraia) host the highest number of alien plants (104, 50 and 56, respectively): these islands also have the highest human population densities, proving that human influence on the islands is the major determinant for alien species abundance. This fact is confirmed by the case of Montecristo, currently an uninhabited island, which hosts the lowest number of alien species, despite being medium-sized. These data are consistent with the findings by Pretto et al. (2012) on small Mediterranean islands, and in other studies (see Heatwole and Walker 1989; McMaster 2005; Kueffer et al. 2010) which highlight the importance of human pressure on the richness of non-native flora on islands.

The alien flora of the Tuscan Archipelago is characterized by a high taxonomic diversity. Most families counted very few species, with only Asteraceae and Poaceae being well-represented. This reflects the non-natural origin of the flora, comprising species from different home-ranges, which did not have the time to evolve into a natural assemblage.

In the Tuscan Archipelago a change in the introduction pathways from agricultural to ornamental purposes was detected as a consequence of the recent socio-economic variation from a historical land-use based on agriculture, livestock and forest exploitation to a model based on tourism, leading to a huge increase in the number of alien taxa. Many other studies are needed to understand the models of arrival, stabilization and modification of the local ecosystems. As stated in the CBD report (CBD decision VI/23 2014),

> invasive alien species represent one of the primary threats to biodiversity, especially in geographically and evolutionarily isolated ecosystems, such as small island developing States, and that risks may be increasing due to increased global trade, transport, tourism and climate change.

Furthermore Mediterranean ecosystems can be seriously endangered by biological invasion (Underwood et al. 2009), especially where their impacts threaten endemic species (Foggi 2005). Hence, strong enforcement of laws and regulations and an intensive education and outreach effort are needed to prevent further introductions of invasive plants in the island biota.

If the local land managers are not able to drive the changes occurring in society, arrival and establishment of alien plants, such as ornamental species, will lead to such changes in landscapes of these islands that they will no longer be recognizable from any other island scattered in the seas of the world.

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Appendix 1. Checklist of the spontaneous alien plants of the Tuscan Archipelago, additional information.

For each alien taxa are given the Raunkiaer's life form (Ph = phanerophytes, nP =nano-phanerophytes, Ch= chamaephytes, H = hemycryptophytes, G = geophytes, Th = therophytes), the introduction pathway, the native range (N = north; C = central; S = south; T = tropical), the residence time (Arch = archaeophytes, Neo = neophytes) and the invasion status (INV = invasive, NAT = naturalized, CAS = casual, cult = only cultivated and nr = not retrieved on the island after 1950) for the whole Archipelago, the seven main islands and for the islets (ARC = whole Archipelago, Cap = Capraia, Elb = Elba, Gia = Giannutri, Gig = Giglio, Gor = Gorgona, Mon = Montecristo, Pia = Pianosa, Isl = islets).

					Invasion status								
Taxon	**Life form**	**Introduction pathway**	**Native Range**	**Res. Time**	**ARC**	**Cap**	**Elb**	**Gia**	**Gig**	**Gor**	**Mon**	**Pia**	**Isl**
Abutilon theophrasti Medik.	Th	Unintentional	Asia	Arch	CAS					CAS			
Acacia dealbata Link	Ph	Forestry, Ornamental	Australia	Neo	INV		INV		NAT				
Acacia melanoxylon R. Br.	Ph	Ornamental	Australia	Neo	NAT	CAS	NAT						
Acacia provincialis A. Camus	Ph	Ornamental	Australia	Neo	CAS		CAS		cult				CAS
Acacia pycnantha Benth.	Ph	Forestry, Ornamental	Australia	Neo	INV		INV						
Acacia saligna (Labill.) H.L. Wendl.	Ph	Ornamental	Australia	Neo	CAS		CAS	cult					
Aeonium arboreum (L.) Webb & Berthel.	nP	Ornamental	Africa	Neo	NAT	CAS	CAS	CAS	NAT	CAS		NAT	
Aeonium haworthii Salm-Dyck ex Webb & Berth.	nP	Ornamental	Africa	Neo	CAS				CAS			CAS	
Agave americana L.	Ph	Ornamental	NC America	Neo	INV	INV	INV	INV	INV	NAT	NAT	INV	NAT
Ailanthus altissima (Mill.) Swingle	Ph	Forestry, Ornamental	Asia	Neo	INV	INV	INV			INV	INV	INV	
Aloe arborescens Mill.	nP	Ornamental	Africa	Neo	CAS	cult	CAS	cult	cult	cult	cult	cult	
Amaranthus albus L.	Th	Unintentional	N America	Neo	NAT	NAT	NAT		nr	NAT	nr	nr	
Amaranthus blitoides S. Watson	Th	Unintentional	N America	Neo	CAS		CAS					CAS	
Amaranthus blitum L.	Th	Unintentional	N America	Neo	CAS	nr	nr			CAS			
Amaranthus caudatus L.	Th	Ornamental	S America	Neo	CAS		CAS						
Amaranthus deflexus L.	H	Unintentional	S America	Neo	INV	INV	INV		INV	NAT	nr	NAT	NAT
Amaranthus graecizans L.	Th	Unintentional	T Africa - T Asia	Neo	NAT		NAT		nr	CAS	CAS		
Amaranthus hybridus L.	Th	Unintentional	NC America	Neo	CAS	CAS	nr			CAS			
Amaranthus retroflexus L.	Th	Unintentional	CS America	Neo	INV	INV	INV	NAT	INV	NAT	nr	INV	
Anredera cordifolia (Ten.) Steenis	G	Ornamental	S America	Neo	INV	NAT	NAT		NAT	NAT		INV	
Antirrhinum majus L. subsp. majus	Ch	Ornamental	Mediterranean	Arch	NAT		NAT		NAT			NAT	
Artemisia verlotiorum Lamotte	H	Unintentional	Asia	Neo	NAT		NAT						
Arundo donax L.	G	Agriculture	Eurasia	Arch	INV	INV	INV	NAT	INV	NAT	nr	INV	NAT
Asclepias fruticosa L.	Ph	Agriculture	Africa	Neo	NAT		CAS		NAT				
Asparagus falcatus L.	G	Ornamental	S Africa	Neo	CAS		CAS						
Austrocylindropuntia subulata	Ch	Ornamental	S America	Neo	NAT	NAT	NAT	CAS	CAS		CAS	NAT	

(*Continued*)

Appendix 1. (*Continued***).**

					Invasion status								
Taxon	**Life form**	**Introduction pathway**	**Native Range**	**Res. Time**	**ARC**	**Cap**	**Elb**	**Gia**	**Gig**	**Gor**	**Mon**	**Pia**	**Isl**
(Muehlenpf.) Backeb.													
Avena byzantina K. Koch	Th	Agriculture	Mediterranean	Arch	CAS					CAS			
Avena sativa L. s.l.	Th	Agriculture	Mediterranean	Arch	CAS				CAS	cult		CAS	
Bidens pilosus L.	Th	Unintentional	CS America	Neo	NAT		NAT						
Brassica napus L. subsp. napus	H	Agriculture	Mediterranean	Arch	CAS		nr			CAS		CAS	
Brassica oleracea L.	Ch	Agriculture	Mediterranean	Arch	CAS	CAS	nr			CAS			
Broussonetia papyrifera (L.) Vent.	Ph	Ornamental	Asia	Neo	NAT	CAS						NAT	
Buddleja davidii Franch.	nP	Ornamental	Asia	Neo	NAT		NAT						
Callitropsis glabra (Sudw.) D.P. Little	Ph	Forestry, Ornamental	NC America	Neo	NAT		NAT	cult	cult				
Campsis radicans (L.) Bureau	Ph	Ornamental	N America	Neo	NAT			cult	NAT		CAS		
Canna indica L.	G	Ornamental	CS America	Neo	CAS		CAS	cult					
Carpobrotus acinaciformis (L.) L. Bolus	Ch	Ornamental	S Africa	Neo	INV	INV	INV	INV	INV	INV	CAS		NAT
Carpobrotus edulis (L.) N.E. Br.	Ch	Ornamental	S Africa	Neo	INV		INV	INV	INV				
Casuarina equisetifolia L.	Ph	Ornamental	Australia	Neo	NAT							NAT	
Catharanthus roseus (L.) G. Don	Th	Ornamental	Africa	Neo	NAT		NAT						
Cenchrus longisetus M.C. Johnst.	Th	Ornamental	Africa	Neo	CAS		CAS						
Chamaesyce humifusa (Willd. ex Schltr.) Prokh.	Th	Unintentional	Asia	Neo	NAT		NAT						
Chamaesyce maculata (L.) Small	Th	Unintentional	N America	Neo	INV	INV	INV					INV	
Chamaesyce prostrata (Aiton) Small	Th	Unintentional	N America	Neo	NAT			NAT	NAT				
Chasmanthe aethiopica (L.) N.E. Br.	G	Ornamental	S Africa	Neo	NAT	NAT	CAS						
Citrullus lanatus (Thunb.) Matsum. & Nakai	Th	Agriculture	Africa	Arch	CAS		CAS	CAS		CAS			
Cortaderia selloana (Schult. & Schult. f.) Asch. & Graebn.	H	Ornamental	S America	Neo	INV	cult	INV			cult			
Cotoneaster coriaceus Franch.	nP	Ornamental	Asia	Neo	NAT		NAT						
Cotyledon orbiculata L.	nP	Ornamental	Africa	Neo	CAS	CAS	CAS	CAS	CAS			CAS	
Crassula lycopodioides Lam.	Ch	Ornamental	S Africa	Neo	CAS		CAS	CAS	CAS				
Crataegus azarolus L.	Ph	Agriculture, Ornamental	Mediterranean	Arch	NAT	NAT							
Cucurbita maxima Duchesne	Th	Agriculture	S America	Neo	CAS		CAS			CAS			

(*Continued*)

Appendix 1. (*Continued*).

Taxon	Life form	Introduction pathway	Native Range	Res. Time	Invasion status ARC	Cap	Elb	Gia	Gig	Gor	Mon	Pia	Isl
Cupressus sempervirens L.	Ph	Forestry, Ornamental	Mediterranean	Arch	NAT	CAS	NAT	cult	NAT	CAS	CAS	NAT	
Cuscuta epilinum Weihe	Th	Unintentional	Asia	Arch	CAS	CAS	nr			nr			
Cyperus involucratus Rottb.	H	Ornamental	Africa	Neo	NAT		NAT						
Datura ferox L.	Th	Unintentional	Asia	Neo	NAT							NAT	
Datura stramonium L. subsp. Stramonium	Th	Unintentional	N America	Neo	NAT	NAT	NAT		nr	NAT	nr	NAT	
Datura wrigthii Regel	Th	Ornamental	N America	Neo	NAT	NAT		CAS	cult				
Delairea odorata Lem.	Ch	Ornamental	S Africa	Neo	NAT		NAT						
Delosperma cooperi (Hook. f.) L. Bolus	H	Ornamental	S Africa	Neo	CAS	CAS			cult			cult	
Drosanthemum floribundum (Haw.) Schwantes	Ch	Ornamental	S Africa	Neo	NAT		CAS		NAT				
Dysphania ambrosioides (L.) Mosyakin & Clemants	Th	Unintentional	S America	Neo	INV	INV	NAT						
Dysphania multifida (L.) Mosyakin & Clemants	H	Unintentional	S America	Neo	NAT				NAT				
Eleusine indica (L.) Gaertn. subsp. indica	Th	Unintentional	T Africa - T Asia	Neo	NAT							NAT	
Erigeron bonariensis L.	Th	Unintentional	CS America	Neo	INV	INV	INV	INV	INV	INV	NAT	INV	NAT
Erigeron canadensis L.	Th	Unintentional	N America	Neo	NAT	NAT	nr	nr	nr	nr	nr	nr	
Erigeron karvinskianus DC.	H	Ornamental	N America	Neo	INV		INV						
Erigeron sumatrensis Retz.	Th	Unintentional	N america	Neo	INV	INV	INV	NAT	INV	NAT	NAT	INV	
Erysimum cheiri (L.) Crantz	Ch	Ornamental	Mediterranean	Arch	NAT	NAT	NAT		NAT	NAT		NAT	
Erythrostemon gilliesii (Wall. ex Hook.) Klotzsch	Ph	Ornamental	S America	Neo	CAS		CAS						
Eucalyptus camaldulensis Dehnh.	Ph	Forestry, Ornamental	Australia	Neo	NAT	CAS	NAT	cult	NAT			NAT	
Eucalyptus globulus Labill.	Ph	Forestry, Ornamental	Australia	Neo	NAT	cult	cult		cult		NAT		
Fallopia baldschuanica (Regel) Holub	Ph	Ornamental	Asia	Neo	NAT		NAT						
Freesia alba (G.L. Mey.) Gumbl.	G	Ornamental	S Africa	Neo	CAS	CAS							
Gazania linearis (Thumb.) Druce	H	Ornamental	S Africa	Neo	CAS		CAS						
Gleditsia triacanthos L.	Ph	Ornamental	N America	Neo	CAS		CAS						
Helianthus annuus L.	Th	Agriculture	N america	Neo	CAS							CAS	
Helianthus tuberosus L.	G	Agriculture, Ornamental	N america	Neo	NAT		NAT			CAS			
Ipomoea indica (Burm.) Merr.	G	Ornamental	S America	Neo	NAT		NAT						

(*Continued*)

Appendix 1. (*Continued*).

					Invasion status								
Taxon	**Life form**	**Introduction pathway**	**Native Range**	**Res. Time**	**ARC**	**Cap**	**Elb**	**Gia**	**Gig**	**Gor**	**Mon**	**Pia**	**Isl**
Iris germanica L.	G	Ornamental	Eurasia	Arch	NAT	NAT	NAT	NAT	NAT	CAS	CAS	NAT	
Kalanchoë × houghtonii D.B.Ward	nP	Ornamental	Africa	Neo	NAT		CAS	NAT				NAT	
Lantana camara L.	Ph	Ornamental	CS America	Neo	NAT	cult		CAS	NAT				
Lepidium didymum L.	Th	Unintentional	N America	Neo	NAT		NAT				NAT		
Ligustrum lucidum W.T. Aiton	Ph	Ornamental	Asia	Neo	NAT		NAT						
Lonicera japonica Thunb.	Ph	Ornamental	Asia	Neo	NAT		NAT						
Lupinus albus L. subsp. albus	Th	Agriculture	Mediterranean	Arch	CAS		nr		nr			CAS	
Melia azedarach L.	Ph	Ornamental	Asia	Neo	CAS	CAS							
Mesembryanthemum cordifolium L. f.	Ch	Ornamental	S Africa	Neo	INV		NAT	INV	NAT	NAT		NAT	
Mirabilis jalapa L.	G	Ornamental	S America	Neo	INV	INV	INV		NAT	NAT		INV	
Morus alba L.	Ph	Agriculture, Ornamental	Asia	Arch	CAS	CAS	CAS			cult	cult	nr	
Myoporum insulare R. Br.	Ph	Ornamental	Australia	Neo	CAS		CAS	CAS					
Nicotiana glauca Graham	nP	Ornamental	S America	Neo	INV	INV							
Nothoscordum gracile (Aiton) Stearn	G	Ornamental	C America	Neo	CAS							CAS	
Opuntia dillenii (Ker-Gawl.) Haw.	nP	Ornamental	S America	Neo	NAT				NAT				
Opuntia ficus-indica (L.) Mill.	Ph	Agriculture, Ornamental	C America	Neo	INV	INV	INV	INV	INV	NAT		NAT	
Opuntia leucotricha DC.	Ph	Ornamental	C America	Neo	NAT		NAT						
Opuntia monacantha (Willd.) Haw.	Ph	Ornamental	S America	Neo	NAT	cult	NAT	NAT			NAT	CAS	
Opuntia phaeacantha Engelm.	Ph	Ornamental	NC America	Neo	INV			INV					
Opuntia stricta (Haw.) Haw.	nP	Ornamental	C America	Neo	INV	INV			INV			nr	
Osteospermum ecklonis (DC.) Norl.	H/Ph	Ornamental	S Africa	Neo	CAS	CAS	CAS	cult	cult				
Oxalis articulata Savigny	G	Ornamental	S America	Neo	NAT	NAT	NAT		NAT	CAS	NAT	NAT	
Oxalis bowiei Herb. ex Lindl.	G	Ornamental	S Africa	Neo	NAT		NAT						
Oxalis debilis Kunth	G	Unintentional	S America	Neo	NAT		NAT						
Oxalis dillenii Jacq.	H	Unintentional	N America	Neo	NAT		NAT	NAT					
Oxalis latifolia Kunth	G	Unintentional	CS America	Neo	NAT		NAT		NAT				
Oxalis pes-caprae L.	G	Ornamental	S Africa	Neo	INV	INV	INV	INV	INV	NAT	NAT	INV	
Paraserianthes lophantha (Willd.) I.C. Nielsen	Ph	Ornamental	Australia	Neo	NAT	NAT	NAT						
Parthenocissus quinquefolia (L.) Planch.	Ph	Ornamental	N America	Neo	NAT		NAT	NAT	NAT			NAT	
Paspalum dilatatum Poir.	H	Unintentional	S America	Neo	NAT		NAT		NAT				
Paspalum distichum L.	G	Unintentional	S America	Neo	NAT	NAT	NAT		NAT	NAT			
Passiflora caerulea L.	Ph	Ornamental	S America	Neo	NAT	cult	NAT						

(*Continued*)

Appendix 1. (*Continued*).

Taxon	Life form	Introduction pathway	Native Range	Res. Time	Invasion status								
					ARC	Cap	Elb	Gia	Gig	Gor	Mon	Pia	Isl
Pelargonium zonale (L.) L'Hér.	Ch	Ornamental	S Africa	Neo	CAS	CAS	CAS	CAS	CAS			CAS	
Phacelia tanacetifolia Benth.	Th	Ornamental	NC America	Neo	CAS					CAS			
Phoenix canariensis Chabaud	Ph	Ornamental	Canary Islands	Neo	CAS		CAS	cult	cult				
Phytolacca americana L.	H	Ornamental	N America	Neo	INV		INV		NAT	NAT	NAT		
Pinus canariensis C. Sm.	Ph	Forestry, Ornamental	Canary Islands	Neo	NAT		NAT		cult				
Pisum sativum L. subsp. sativum	Th	Agriculture	Eurasia	Arch	CAS	nr	cult		CAS			CAS	
Pittosporum tobira (Thunb.) W.T. Aiton	Ph	Ornamental	Asia	Neo	NAT	CAS	NAT	CAS	CAS				
Prunus armeniaca L.	Ph	Agriculture	Asia	Arch	CAS	cult	CAS		cult		cult	CAS	
Prunus cerasus L.	Ph	Agriculture	Eurasia	Arch	CAS	CAS	CAS		CAS			CAS	
Prunus domestica L. s.l.	Ph	Agriculture	Eurasia	Arch	NAT	NAT	nr		CAS				
Robinia pseudoacacia L.	Ph	Forestry, Ornamental	N America	Neo	INV	INV	INV		INV	INV	cult	INV	
Senecio angulatus L. f.	H	Ornamental	S Africa	Neo	INV	INV	INV	INV	INV	NAT		INV	
Setaria parviflora (Poir.) Kerguélen	H	Unintentional	S America	Neo	NAT		NAT						
Solanum lycopersicum L.	Th	Agriculture	C America	Neo	CAS	CAS	CAS	CAS	CAS	CAS	cult	CAS	
Sorghum bicolor (L.) Moench	Th	Agriculture	Africa	Arch	CAS	cult				CAS			
Sorghum halepense (L.) Pers.	G	Unintentional	Mediterranean	Arch	INV	NAT	INV	NAT			nr	NAT	
Symphyotrichum squamatum (Spreng.) G.L. Nesom	Th	Unintentional	S America	Neo	INV	INV	INV	NAT		NAT		INV	
Tamarix parviflora DC.	Ph	Ornamental	Mediterranean	Neo	CAS		CAS		cult				
Tanacetum parthenium (L.) Sch. Bip.	H	Ornamental	Eurasia	Arch	CAS		nr		CAS				
Tetragonia tetragonoides (Pallas) Kuntze	Th	Agriculture	Australia	Neo	CAS			CAS					
Triticum aestivum L.	Th	Agriculture	Asia	Arch	CAS		CAS				CAS	CAS	
Tropaeolum majus L.	Th	Ornamental	S America	Neo	CAS	CAS	CAS	cult					
Vachellia karroo (Hayne) Banfi & Galasso	Ph	Ornamental	S Africa	Neo	CAS		CAS						
Veronica persica Poir.	Th	Unintentional	Asia	Neo	INV	NAT	INV			NAT			
Vitis riparia Michx. × *Vitis rupestris* Scheele	Ph	Agriculture	N America	Neo	INV	NAT	INV	cult	NAT	cult		cult	
Xanthium orientale L. subsp. italicum (Moretti) Greuter	Th	Unintentional	N America	Neo	NAT	NAT	NAT		NAT			CAS	
Xanthium spinosum L.	Th	Unintentional	S America	Neo	NAT		NAT		nr	NAT	nr		
Yucca aloifolia L.	Ph	Ornamental	NC America	Neo	CAS	cult	CAS	cult	cult		cult		
Zantedeschia aethiopica (L.) Spreng.	G	Ornamental	S Africa	Neo	INV	INV	CAS						
Ziziphus jujuba Mill.	Ph	Agriculture	Asia	Arch	NAT		NAT					NAT	

4. IMPACTS ON THE ECOSYSTEM

The evaluation of impacts of IAS on biodiversity is a main goal of invasion biology. Indeed quantitative information on impacts exerted by alien plant are very important to fully understand biological invasions. Moreover, quantitative assessment of alien plant impacts are essential in the management of biological invasions, being indispensable to prioritize the resources against the most problematic species and restoration efforts towards the worst-affected ecosystem processes (Hulme et al., 2013). They are in fact a central pillar in most Risk Assessment procedures, allowing for prioritizing alien species according to their impact (Essl et al., 2011). Thus, further research on the impacts of alien species on native biodiversity is needed to both achieve a better understanding of this phenomenon and to acquire more data for Risk Assessments (Essl et al., 2011).

Hulme et al. (2013) emphasized the need of unbiased researches on impacts, identifying gaps resulting from taxonomic, biogeographic, and life-form biases and from the lack of comprehensive approaches, investigating impacts larger sets of response variables as well as the consequences upon ecosystem services. On the one hand, the authors highlighted that most of the researches focused on few well-studied species, while data on poorly study species are essential. On the other hand most of the impact studies examine one, or few variables, often not integrated, while more comprehensive approaches relating the effects of alien plants to more responses, also focusing to the less studied, such as impacts on soil biota or ecosystem stocks. Moreover, studies on impacts should be more strongly linked to invasive plant functional traits and their relationship with ecosystem processes.

These biases are also reflected in the problems faced by land managers when assessing the potential or actual risks posed by invasive alien plant species on Mediterranean Islands (Brundu, 2013) Particularly when facing species that are scarcely studied or which complex of impacts are poorly understood or investigate specifically on islands.

In our area of study, an evaluation of quantitative assessment of impacts exerted by invasive species was still lacking, also for well-known species. Thus, aiming to better understand the ecological consequences of invasion in Tuscan Archipelago, we investigate, as an example, the effect on biodiversity of two alien species. We produced two contribution focused on the impacts of two Australian Acacias species: *Acacia dealbata* Link and *Acacia pycnantha* Benth, which were offering the opportunity to assess quite complex and effective changes in the invaded ecosystems.

Generally, Australian acacias are a group of globally introduced species including at least 23 well known or emerging invaders in many part of the world, especially in Mediterranean ecosystems, where lead to a wide range of ecological and socio-economic impacts (Lorenzo et al., 2010; Le Maitre et al. 2011). Acacia's spread and dominance can lead to diversity loss, alteration of functional diversity and simplification of invaded habitats. The severe impacts of acacias are related to some key traits such as the high growth rate and biomass accumulation, the production of a large and persistent seed bank, the capacity to establish associations with nitrogen-fixing bacteria (Le Maitre et al. 2011), and the release of allelopathic compounds (Lorenzo et al. 2010). Particularly to our studied species, the former, *A. dealbata*, is a well-studied species, whose effect on biodiversity and nutrient cycles have been largely studied, but still never in an island ecosystem (Fuentes-Ramírez and Pauchard 2010; González-Muñoz et al. 2012; Lorenzo et al. 2012). Moreover, to describe the effect of this species we adopted a comprehensive approach, assessing its effects on plant communities as well as on soil chemical properties and soil microbial communities (i.e bacterial and fungal). The second species, *A. pycnantha*, is instead a known invader, but whose impacts have benne scarcely studied, and for which quantitative information on ecological impacts are still lacking.

In Tuscan Archipelago, and particularly on the Island of Elba, Australian acacias were sporadically introduced in pine tree plantation (Gatteschi and Arretini, 1990). Particularly, in the area of study of these two contribution, located in the South-East of the Island of Elba (Figure 4.1), both the two species were largely used. In 1998, a large fire affected the area, destroying entirely the pine tree plantation and the local vegetation. As a result, the two acacias species, which as most of the acacia are facilitate by fire, spread across the area, replacing the native sclerouphyllous scrubland that would have colonized the area. Nowadays these two acacias form several dense monospecific stands, rising to around 30 hectares of total surface, quite well observable and different from surrounding native shrubland (Fig 4.1)

Given the spotty spatial distribution of the invasion, to investigate the effects of these two species on the ecosystem, we adopted a hierarchical sampling of the communities. The sampling was conducted in spring 2013 with similar sampling method for the two species. In case of *A. dealbata* we sampled and analyzed understory vegetation; soil chemical properties and soil microbial communities, while in case of *A. pycnantha* only understory plant communities were taken into account.

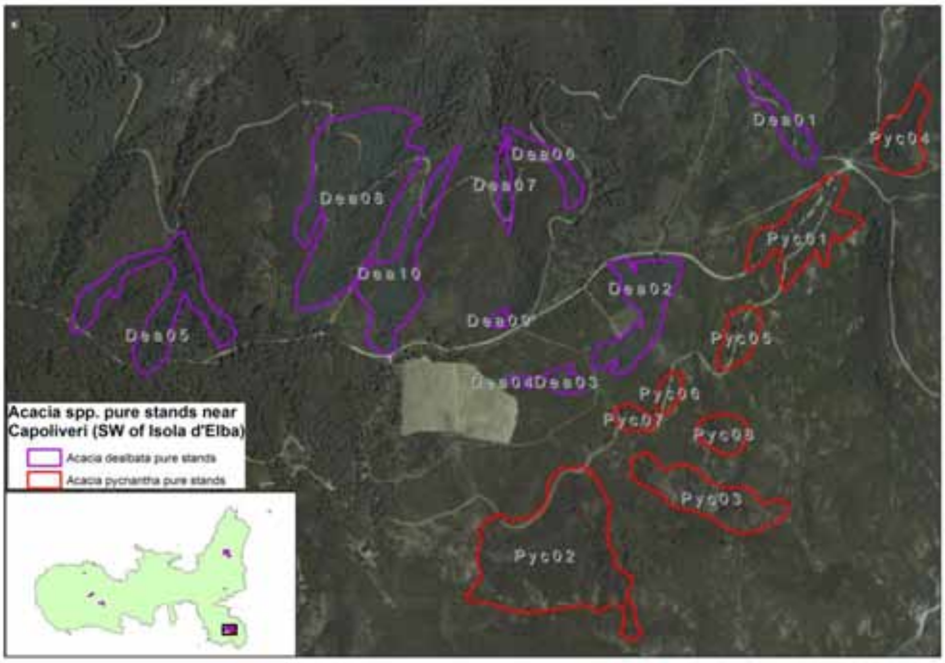

Figure 4.1 Area of study of the two contribution on the ecological impacts of *Acacia dealbata* and *A. pycnantha*. In the map, the invaded patches are marked with different color according to the species.

For each species, we selected three patches, and in each of these, we selected two transects going from invaded to non-invaded vegetation, throughout a transitional invasion stage. Finally along the transect several subplot were sampled per invasion status. The invasion statuses were defined as follow: (1) 'invaded' with the vegetation dominated by *A. dealbata*, (2) 'non-invaded', consisting of contiguous native understory communities without any *A. dealbata* individuals, and (3) 'transition' where an intermediate degree of invasion was detected.

The data coming from the sampling were analyzed adopting univariate and multivariate statistics, studying the variation of several responses across the three statuses of invasion. Particularly in both the analyses we took in consideration the hierarchical structure of the data. In case of univariate analyses we used the framework of multilevel Generalized Linear Mixed Models (GLMM). Adopting this kind of analysis, with the specification of random factor (i.e. highest hierarchical levels such as “patch” and “transect”) allowed us to study variation in subplot disregarding from the results from local variability leading to more general conclusions (Bolker et al., 2009). Also in the multivariate analyses the hierarchical structure of the data was taken into account, using the “transect” as a covariate in the analyses and constraining the permutation used in these analyses to run according to the hierarchical structure of the dataset.

These work led in the beginning to the production of a poster in the EMAPI 2013 congress (12th Reunion on ecology and management of alien plant invasions; Pirenópolis, Goiás, Brazil; 22-26 September 2013), illustrating the preliminary data coming from the case of *A. dealbata* (Section 4.1). Finally two main contribution were realized, one regarding impacts of *A. dealbata* published in 2014 (Section 4.2); and a second regarding impacts of *A pycnantha*, resubmitted after major revision requested (Section 4.3).

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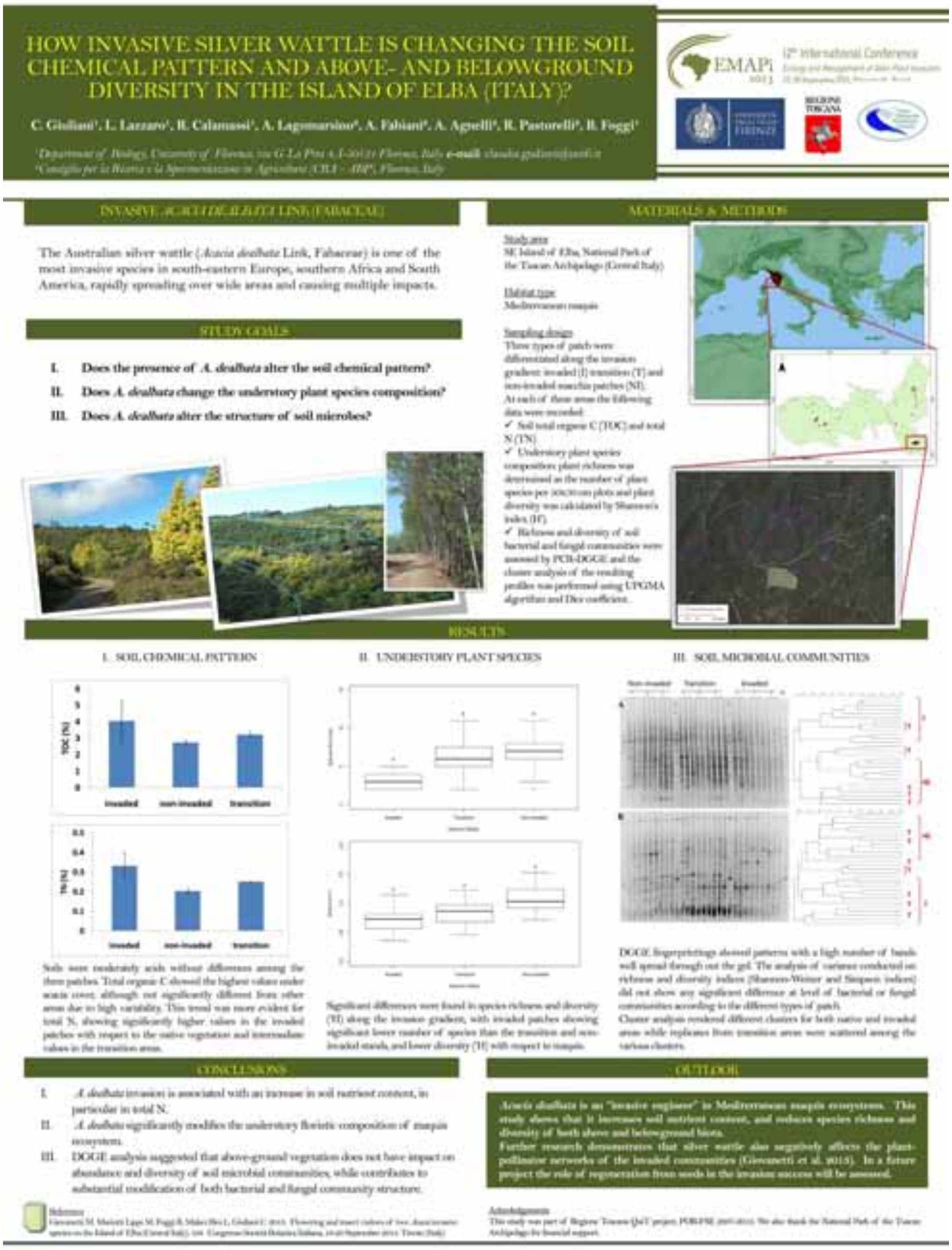
4.1. How invasive silver wattle is changing the soil chemical pattern and above- and below-ground diversity in the Island of Elba (Italy)?



HOW INVASIVE SILVER WATTLE IS CHANGING THE SOIL CHEMICAL PATTERN AND ABOVE- AND BELOWGROUND DIVERSITY IN THE ISLAND OF ELBA (ITALY)?

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INVASIVE *ACACIA DEALBATA* LINK (FABACEAE)

The Australian silver wattle (*Acacia dealbata* Link, Fabaceae) is one of the most invasive species in south-eastern Europe, southern Africa and South America, rapidly spreading over wide areas and causing multiple impacts.

STUDY GOALS

I. **Does the presence of *A. dealbata* alter the soil chemical pattern?**
II. **Does *A. dealbata* change the understory plant species composition?**
III. **Does *A. dealbata* alter the structure of soil microbes?**

MATERIALS & METHODS

Study area
SE Island of Elba, National Park of the Tuscan Archipelago (Central Italy)

Habitat type
Mediterranean maquis

Sampling design
Three types of patch were differentiated along the invasion gradient: invaded (I) transition (T) and non-invaded maquis patches (NI).
At each of these areas the following data were recorded:
✓ Soil total organic C (TOC) and total N (TN)
✓ Understory plant species composition: plant richness was determined as the number of plant species per 30x30 cm plots and plant diversity was calculated by Shannon's index (H').
✓ Richness and diversity of soil bacterial and fungal communities were assessed by PCR-DGGE and the cluster analysis of the resulting profiles was performed using UPGMA algorithm and Dice coefficient.

RESULTS

I. SOIL CHEMICAL PATTERN

Soils were moderately acids without differences among the three patches. Total organic C showed the highest values under acacia cover, although not significantly different from other areas due to high variability. This trend was more evident for total N, showing significantly higher values in the invaded patches with respect to the native vegetation and intermediate values in the transition areas.

II. UNDERSTORY PLANT SPECIES

Significant differences were found in species richness and diversity (H') along the invasion gradient, with invaded patches showing significant lower number of species than the transition and non-invaded stands, and lower diversity (H') with respect to maquis.

III. SOIL MICROBIAL COMMUNITIES

DGGE fingerprintings showed patterns with a high number of bands well spread through out the gel. The analysis of variance conducted on richness and diversity indices (Shannon-Weiner and Simpson indices) did not show any significant difference at level of bacterial or fungal communities according to the different types of patch.
Cluster analysis rendered different clusters for both native and invaded areas while replicates from transition areas were scattered among the various clusters.

CONCLUSIONS

I. *A. dealbata* invasion is associated with an increase in soil nutrient content, in particular in total N.
II. *A. dealbata* significantly modifies the understory floristic composition of maquis ecosystem.
III. DGGE analysis suggested that above-ground vegetation does not have impact on abundance and diversity of soil microbial communities, while contributes to substantial modification of both bacterial and fungal community structure.

OUTLOOK

Acacia dealbata is an "invasive engineer" in Mediterranean maquis ecosystems. This study shows that it increases soil nutrient content, and reduces species richness and diversity of both above and belowground biota.
Further research demonstrates that silver wattle also negatively affects the plant-pollinator networks of the invaded communities (Giovanetti et al. 2013). In a future project the role of regeneration from seeds in the invasion success will be assessed.

Reference
[illegible]

Acknowledgements
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4.2 Soil and plant changing after invasion: The case of *Acacia dealbata* in a Mediterranean ecosystem

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Soil and plant changing after invasion: The case of *Acacia dealbata* in a Mediterranean ecosystem

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HIGHLIGHTS

- We analyzed the impacts of *A. dealbata* invasion on a Mediterranean ecosystem.
- We analyzed impacts on soils, soil bacterial, fungal and plant communities.
- Acidification, nitrification and a slight increase in C were detected for soils.
- A compositional shift was detected for bacterial and fungal communities.
- Lower richness and functional shift were detected for understory plant communities.

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ABSTRACT

Acacia dealbata Link (Fabaceae) is one of the most invasive species in the Mediterranean ecosystems of Europe, Africa and America, where it has been proved to exert strong effects on soil and plant communities. In Italy *A. dealbata* has been largely used for ornamental and forestry purpose and is nowadays spreading in several areas. The present study was addressed to evaluate the impacts on soil chemical properties, soil microbial communities and understory plant communities and to assess the relationships among these compartments after the invasion of *A. dealbata* in a typical Mediterranean shrubland.

Towards these aims, a soil and vegetation sampling was performed in Elba Island where *A. dealbata* is invading the sclerophyllous native vegetation. Three levels of invasion status were differentiated according to the gradient from invaded, to transitional and non-invaded vegetation.

Quantitative and qualitative alterations of soil chemical properties and microbial communities (i.e. bacterial and fungal communities) and above-ground understory plant communities were found. In particular, the invaded soils had lower pH values than both the non-invaded and transitional ones. High differences were detected for both the total N and the inorganic fraction (NH_4^+ and NO_3^-) contents, which showed the ranking: invaded > transitional > non-invaded soils. TOC and C/N ratio showed respectively higher and lower values in invaded than in non-invaded soils. Total plant covers, species richness and diversity in both the non-invaded and transitional subplots were higher than those in the invaded ones. The contribution of the nitrophilous species was significantly different among the three invasion statuses, with a strong increase going from native to transitional and invaded subplots. All these data confirm that *A. dealbata* modifies several compartments of the invaded ecosystems, from soil chemical properties to soil and plant microbial communities determining strong changes in the local ecosystem processes.

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1. Introduction

Impacts related to invasive alien plants have been recognized as one of the major threats to biodiversity (CBD decision VI/23, 2014) and have been deeply investigated in the last decades (Hulme et al., 2013). However, these studies are often focused on one single or few response variables, while more composite approaches, assessing the effects of invasive plants on several response variables, are needed (Hulme et al., 2013). In particular, investigations on the impacts on soil fauna and microbes and belowground processes are urgently demanded according to their importance in the invasion development (Hulme et al., 2013).

Indeed, invasive plants could alter the quality or quantity of the resources available to soil microorganisms, inducing changes in richness and abundance of particular members of the microbial community with a subsequent alteration of the nutrient pool and dynamics (Jordan et al., 2008; Kao-Kniffin and Balser, 2008), potentially affecting the performance of native plant species.

In this framework, N_2-fixing Australian acacias, that include some of the most important plant invaders on a global scale (Richardson and Rejmánek, 2011), have great influence on nitrogen (N) and carbon (C) dynamics, being able to successfully establish and grow even in resource-limited ecosystems (Funk and Vitousek, 2007).

Within this group, *Acacia dealbata* Link (silver wattle) is a versatile and highly adaptive tree species which has spread all over the world and currently covers a considerable area in the Atlantic climates and Mediterranean regions of Southern Europe, from North-western Iberian Peninsula to Italy (Sheppard et al., 2006), causing huge ecological concerns. In Italy, *A. dealbata* was reported as invasive in the Mediterranean areas of Liguria (Celesti-Grapow et al., 2009) and Tuscany (Elba Island, National Park of the Tuscan Archipelago, Lazzaro et al., 2014), with potential severe conservation consequences in such important hotspot of diversity and endemism (Mittermeier et al., 2004).

The spreading of this transformer species (Richardson et al., 2000) has recently received great attention; serious community level impacts of *A. dealbata* invasion have been documented in North-western Spain and Chile. It enhances nutrient mineralization and decomposition rates (Castro-Díez et al., 2012), modifies the soil microbial communities structure (Lorenzo et al., 2010a), decreases diversity and richness of invertebrate communities (Coetzee et al., 2007), and reduces understory plant richness and diversity (Lorenzo et al., 2012; Fuentes-Ramírez and Pauchard, 2010; González-Muñoz et al., 2012).

Due to the large below- and aboveground transformations induced by *A. dealbata* in the invaded ecosystems, quantitative studies are necessary also in Italy in order to determine whether this invasive species displaces native communities and reduces local biodiversity.

The aim of our work is to assess the impacts of *A. dealbata* on different compartments of a typical Mediterranean sclerophyllous shrubland in Elba Island, exploiting several response variables (i.e. soil chemical properties, soil microbial and understory plant communities). Accordingly, we focused on 3 main questions: (*i*) Does *A. dealbata* qualitatively and quantitatively impact the invaded ecosystems? (*ii*) Do invasion effects affect a wide set of ecosystem compartments? (*iii*) Are these impacts somehow related, i.e. which relationship derived directly or indirectly from *A. dealbata* invasion, and do these occur among the different biotic and abiotic compartments we analyzed? Towards these aims, a sampling of soil and aboveground understory plant communities was conducted across a gradient of invasion spanning from invaded to transitional and non-invaded vegetation.

2. Methods

2.1. Study area

The study took place in areas of ecological interest in the South-eastern part of Elba Island (central Italy), the largest of the seven main islands of the Tuscan Archipelago (Tuscan Archipelago National Park, central Mediterranean Sea). Native vegetation in the study area is a typical Mediterranean vegetation constituted by mixed stands with *Quercus ilex* and *Erica arborea*. *A. dealbata* was introduced in the area in the second half of the 1900s, occasionally used in pine tree plantations (Gatteschi and Arretini, 1990). Nowadays, after a fire in the late 90s several pure stands of *A. dealbata*, ranging from about 1000 square meters to 5 ha of surface, can be detected in the study area, invading the surrounding native vegetation. The study area results homogeneous for the climate (thermo-Mediterranean climate: Foggi et al., 2006) and for the geological substratum (classified as Eutric Cambisols developed on parental material mainly composed of metamorphic rock: white schists; Costantini et al., 2012).

2.2. Sampling design

Three patches invaded by *A. dealbata* were surveyed and for each patch one macroplot was delimited on the basis of the presence of three different degrees of invasion: (1) 'invaded' with the vegetation dominated by *A. dealbata*, (2) 'non-invaded', consisting of contiguous native understory communities without any *A. dealbata* individuals, and (3) 'transition' where an intermediate degree of invasion was detected. For each macroplot, two transects 2 meter wide were randomly selected (at least 20 m apart), going from the invaded, to the transition and the non-invaded vegetation. Soil sampling and vegetation surveys were carried out in middle May 2013.

2.3. Soil sampling and analysis

Soil sampling was carried out in only one of the two transects of each macroplot. The soil samples were collected in three sampling points per each invasion status on each transect. In each sampling point, four cores were collected with a hand auger (5 cm inside diameter) from the 20 cm deep top layer of bulk soil after removing surface residues. The four cores were then pooled together and stored at −20 °C for physico-chemical and microbiological analysis. A total of 27 samples for the soil analyses resulted from the sampling design (three macroplot × one transect × three invasion status × three sampling points).

Soil physical and chemical properties of topsoil were determined according to the analytical methodology approved by the National Observatory for Pedology and Soil Quality (Mipaaf, 2000). The soil has a loam texture (47% sand, 19% clay, 25% fine silt, 10% coarse silt), without differences among the patches. Soil pH was determined in water with an extraction ratio 1:2.5. Total organic C (TOC) and total N (TN) contents in the bulk soil were measured by dry combustion on a Thermo Flash 2000 CN soil analyzer. To this aim, 20 to 40 mg of soil was weighed into Ag-foil capsules. Soil was carbonate free; consequently, a pretreatment with HCl was unnecessary.

NO_3-N and NH_4-N were extracted by shaking 20 g of fresh soil in 100 ml of 2 M KCl solution for 1 h. Soil extracts were analyzed with the FIAstar 5000 Auto Analyzer system (FOSS Höganäs) for NO_3-N and NH_4-N concentration.

2.4. DNA extraction and denaturing gradient gel electrophoresis (DGGE) analysis

DNA was extracted using the FastDNA® SPIN Kit for Soil (MP Biomedicals) following the manufacturer's instructions. The extracted DNA was visualized on an agarose gel (1% w/vol) and DNA yields were estimated by comparison to bacteriophage λ DNA dilutions (200, 100, 50 ng) using the Chemidoc Apparatus (Biorad).

For DGGE analysis, the extracted DNA was amplified using bacterial 16S rDNA (GC986f–UNI1401r; Felske and Akkermans, 1998) and fungal 18S rDNA (EF390–GCFR1; Vainio and Hantula, 2000) specific primers. Amplification reactions were carried out in a MJ Research PTC-200™ thermocycler (Biorad) in a 25-μl mixture containing 2 ng of DNA, 1.5 mmol l^{-1} of $MgCl_2$, 200 μmol l^{-1} of each deoxynucleotide triphosphate (dNTP) (Promega Corporation), 10 pmol of each primer (TIB MolBiol), 1 × reaction buffer (Polymed), and 1 U of PolyTaq polymerase (Polymed), under reaction conditions of 94 °C for 4 min followed by 35 cycles of denaturation at 95 °C for 45 s, annealing (at 55 °C or 48 °C for bacteria and fungi, respectively) for 45 s, extension at 72 °C for 45 s, and final extension at 72 °C for 7 min. Three independent PCR amplifications were performed for each primer set and each soil sample and the triplicate amplification products were pooled to minimize the effect of PCR biases. Amplicon yields were estimated by comparison of amplified DNA to Low DNA mass ladder (Invitrogen) using the Chemidoc Apparatus. DGGE profiles were obtained loading 500 ng of amplicons onto a 6% polyacrylamide gel (acrylamide/bis 37.5:1; Euroclone), with denaturing gradients (ranging from 50 to 65% or from 42 to 52% for 16S and 18S rDNA fragments, respectively) obtained with an 100% denaturing solution containing 40% formamide (Euroclone) and 7 M Urea (Euroclone). The gels were run for 17 h in 1X TAE buffer at constant voltage (80 V) and temperature (60 °C), using the Ingeny phorU-2 System (Ingeny International BV). At the end, gels were stained with SYBR®GOLD (Molecular Probes) diluted 1:1000 in 1X TAE and the gel images digitalized using the Chemidoc Apparatus.

2.5. Soils statistical analysis

The soil physical and chemical properties of the 27 soil samples were studied using GLMMs (generic linear mixed models) using "invasion status" like explanatory variable with fixed effect and the variable "transect" as factor with random effect. If the main factor was significant a Tukey test was used to investigate the differences between the three levels of "invasion status". A log transformation was adopted for the TOC, TN, C/N ratio, NH_4, NO_3. In addition, to assess the changes in soil properties of the samples across the invasion status a Principal Component Analysis (PCA) was performed again using the transects as a covariate to get rid of partial variation due to this grouping factor.

Evaluation of band migration distance and intensity within each lane of the DGGEs was performed using Gel Compare II software v 4.6 (Applied Maths). The number of bands (species richness) and their relative abundance (Shannon index, H') were used as a proxy of richness and diversity of soil microbial communities as described by Pastorelli et al. (2011). Bands with a minimum area below 1% were excluded from the calculations. Differences in richness and Shannon index were studied using GLMMs (with "transect" as a random effect factor). A hierarchical cluster analysis based on position and presence/absence of bands in the different profiles was performed using Dice coefficient and the unweighted pair group method using arithmetic average (UPGMA) algorithm. The banding patterns of DGGEs, extracted as quantitative band matching tables, were further analyzed in a multivariate constrained ordination such as Redundancy Analysis (RDA) to study how the soil chemical properties and the invasion status affected the band composition in the samples. An interactive forward stepwise selection of significant terms, based on adjusted P-values for false discovery rate (FDR as described by Benjamini and Hochberg, 1995) was used to choose the right terms and again transect was used as a covariate. Cluster analysis and band matching were performed again using Gel Compare II while RDA analysis using Canoco 5 for Windows (vers. 5.03).

2.6. Plant communities sampling

Understory plants were sampled on both transects of each macroplot. Along each transect, four 0.5 side meters quadrate subplots were sampled for each different invasion status, where the understory vegetation was sampled. Only herbs, and young shrubs and trees (for these two excluding individuals taller than 30 cm) were included. In each subplot, all plant individuals were recorded (due to the difficulties of determining the real number of individuals of *Brachypodium retusum*, for n > 25 number was estimated). The total plant cover (%) was also estimated as the area covered by all vascular plant species in the quadrate subplot, determined by dividing the quadrate into 100 5 × 5 cm cells and adding the number of occupied cells. A total of 72 subplots (three macroplot × two transects × three invasion status × four quadrate subplots) were sampled.

2.7. Plant communities statistical analysis

The analyses were performed using generic linear mixed models (GLMMs) adopting restricted maximum likelihood method (REML). Species richness, species diversity, expressed by Shannon index H' (Shannon and Weaver, 1949) and total plant cover of the 72 quadrate subplots were used as response variables aiming to investigate how understory plant communities could be affected by *A. dealbata* invasion. The same analysis was used to study the contribution of the guild of nitrophilous species (expressed as percentage on total number of species and relative abundance) defined by the Ellenberg's ecological indicator value for eutrophication (Ellenberg, 2009; Pignatti et al., 2005) above 5. A log transformation was adopted in the case of species richness, while an arcsine transformation was used for total plant cover data and percentage and relative abundance of nitrophilous species (aiming to normalize the residuals). The "invasion status" was used as explanatory variable with fixed effect. The variables "macroplot" and "transect" were used as random effect factors, considering "transect" being nested in "macroplot". If the main factor was significant Tukey test was used to investigate the differences between the three levels of "invasion status".

To explore how the species composition of subplots was affected by different "invasion status" a comparison between a partial-Correspondence Analysis (CA) and a partial-Canonical Correspondences Analysis (CCA) was performed. The variable "transect" was used as a covariate to get rid of partial variation due to this grouping factor, and in the CCA the "invasion status" of the subplots was used as explanatory variable. The variation explained by unconstrained axis is compared to the explained variation by constrained axis to check the efficiency of constrained axis. For the CCA the significance of all constrained axes was tested with a permutation test based on the hierarchical structure of the data, with freely exchangeable permutations run inside each transect.

In the ordinations, *A. dealbata* was excluded from computation while it is part of the definition of the different invasion status. A log transformation was applied to the abundances of species and while the result of this kind of analysis is usually affected by the presence of rare species these were down-weighted in the computations.

The GLMMs for both plant and soils analysis were carried out using the *nlme* package (Pinheiro et al., 2013), and the Tukey tests were carried out exploiting the *multcomp* package (Hothorn et al., 2008) for R software version 3.02 (R Core Team, 2013).

The ordination analyses for both plant and soils analysis were performed using Canoco 5 for Windows (vers. 5.03, Ter Braak and Šmilauer, 2012).

3. Results

According to GLMMs results, pH, TN, NH_4 and NO_3 resulted to be significantly different among the three invasion status ($P < 0.05$, Table 1) whereas marginal significant differences were detected for TOC and C/N ratio ($P < 0.10$, Table 1). The invaded soils had lower pH values than both the non-invaded and transitional ones. High differences were detected for both the total N and the inorganic fractions (NH_4^+ and NO_3^-) contents, which showed the ranking: invaded > transitional > non-invaded soil samples (Table 1). TOC and C/N ratio showed respectively higher and lower values in invaded than non-invaded soils.

The first two axes of the partial-PCA (Fig. 1) of soil chemical variables allowed explaining 74.5% of partial variation (49.4% explained by first axis and 25.1% by second axis). The partial-PCA first axis highlighted a good correlation between the soil properties and the invasion status, with N forms (TN, NH4 and NO3) and TOC positively correlated and pH negatively correlated with the invasion status. According to this interpretation, the soil samples appear to be clustered depending on the status of invasion.

16S-DGGE banding patterns showed marginal significant differences for both species richness and diversity ($P < 0.10$, Table. 1), whereas no significant differences were detected in 18S-DGGE banding patterns (Table. 1). For both 16S and 18S DGGE fingerprints, cluster analysis rendered two different main clusters for the invaded and non-invaded soil samples within which DGGE profiles from transitional areas are scattered (Fig. S-1, Supplemental materials). Furthermore for both 16S and 18S DGGE fingerprints partial-RDA results highlighted that the composition of soil bacterial and fungal communities were highly affected by the invasion status ($P < 0.01$), while they appear to be affected by diverse soil chemical properties (Table 2). The stepwise selection of explanatory variables for partial-RDA on genetic fingerprinting by 16S-DGGE, allowed including the invasion status (decomposed in three dummy variables) and the C/N ratio, which are considered to highly affect the band composition of the soil bacterial communities (Fig. 2 and Table 2). The explanatory variables accounted for 34.1% of partial variation (Partial variation = 4911.171). In the case of 18S-DGGE the forward selection allowed to include in the partial-RDA again the invasion status, the C/N ratio and the NO_3 content (Fig. 3 and Table 2). The explanatory variables accounted for 33.7% of partial variation (PV = 4160.366).

A total of 58 plant species were detected. 14 of these were common to all the three invasion status, whereas 3, 11 and 14 were exclusive to invaded, transition and non-invaded subplots, respectively. Two species were detected in both the invaded and the transition subplots, and 14 in both the transition and the non-invaded ones. No species were shared between the invaded and non-invaded subplots (Table S-1, Supplemental materials).

The GLMMs highlighted significant differences for species richness, diversity, total plant cover and contribution of the nitrophilous species (Table 3). Species richness was higher in the non-invaded and transitional subplots than in the invaded ones. Also diversity was higher in the non-invaded subplots compared to the invaded ones, whereas transitional subplots had slightly higher diversity values than non-invaded subplots. The non-invaded subplots had higher total plant covers than invaded subplots. The contribution of the nitrophilous species was significantly different among the three invasion status, with a strong increase going from native to transitional and invaded ones.

According to the global permutation test on constrained axes of the partial-CCA the subplot species composition was significantly affected by the invasion status (Pseudo F = 2.8, P = 0.0002). The constrained axes of partial-CCA accounted for 8.1% of the partial variation (PV = 3.5744), axis I accounted for 5.6% of total variance, whereas 2.5% of the variation was explained by the second ordination axis (Fig. 4). The comparison with unconstrained axes of the partial-CA highlights the efficiency of the constrained axes in catching the data variation (efficiency of constrained axes was 49.6% for first axis and 25.3% for the second axis). The partial-CA first two axes accounted for 21.2% of total variance (11.3% explained by first axis and 9.9% explained by second axis, see Table S-2 and Fig. S-2, Supplemental materials). Therefore the invasion status highly influenced the species composition of subplots even though the high noise affecting the plant communities.

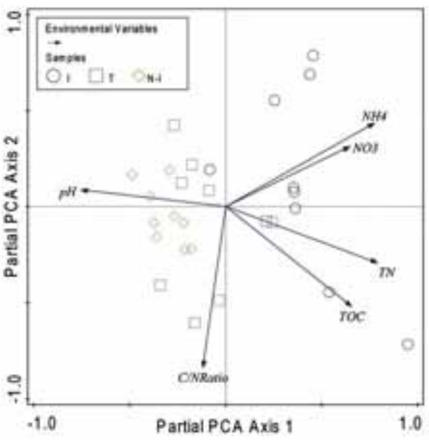

Fig. 1. PCA ordination diagram of soil samples based on chemical properties. TOC = total organic carbon; TN = total nitrogen. Symbols represent the samples grouped according to the invasion status (I = invaded; T = transitional; N-I = non-invaded).

4. Discussion

The aim of the study was to assess how *A. dealbata* invasion is changing above- and belowground biotic and abiotic components of the native ecosystem. We found that *A. dealbata* highly affected almost all the response variables we analyzed, thus impacting both abiotic and biotic compartments of the native ecosystem and acting through different ecological processes. Silver wattle invasion directly impacted soil properties through a general improvement in soil N availability, a higher litter layer accumulation leading to marginally higher TOC, and soil acidification. In a wide variety of habitats, invasive plant species are shown to be able to exert strong effects on soil ecosystem functioning and dynamics, altering the spatial distribution of nutrients, and our

Table 1
Results for the GLMMs on soil chemical properties and microbial communities of the topsoil (0–20 cm). Means (n = 9) calculated by invasion status ± standard errors are shown. Entries in bold highlight significant statistical differnces at P = 0.05. Different letters in a column indicate significant differences at P < 0.05 between means, while letters in round parenthesis indicate marginally significant differences at P = 0.10 according to the Post-hoc test (TOC, total organic carbon; TN, total nitrogen).

Variable	Invaded	Transition	Non-invaded	nDF	dDF	F-value	P-value
pH	**5.05 ± 0.14 a**	**5.66 ± 0.11 b**	**5.66 ± 0.11 b**	**2**	**22**	**8.56**	**0.0018**
TOC ($g\ kg^{-1}$)	40.56 ± 6.91 b	32.22 ± 1.59 ab	27.45 ± 1.77 a	2	22	3.2906	0.0562
TN ($g\ kg^{-1}$)	**3.32 ± 0.40 c**	**2.51 ± 0.07 b**	**2.04 ± 0.11 a**	**2**	**22**	**14.76**	**0.0001**
C/N ratio	11.77 ± 0.59 (a)	12.83 ± 0.42 (ab)	13.57 ± 0.71 (b)	2	22	2.602	0.0968
NH_4 ($mg\ kg^{-1}$)	**15.21 ± 2.21 c**	**6.18 ± 1.77 b**	**1.10 ± 0.81 a**	**2**	**22**	**27.455**	**<.0001**
NO_3 ($mg\ kg^{-1}$)	**4.28 ± 1.03 c**	**2.15 ± 0.93 b**	**0.09 ± 0.09 a**	**2**	**22**	**24.088**	**<.0001**
X16S.Richness	19.78 ± 0.74 (a)	22.22 ± 0.76 (ab)	22.78 ± 1.32 (b)	2	22	2.84	0.0801
X18S.Richness	12.11 ± 0.93	12 ± 1.08	12.11 ± 1.18	2	22	0.01	0.9859
X16S.Shannon	2.94 ± 0.04 (a)	3.06 ± 0.03 (ab)	3.08 ± 0.06 (b)	2	22	2.87	0.0782
X18S.Shannon	2.43 ± 0.08	2.4 ± 0.09	2.41 ± 0.1	2	22	0.03	0.9679

Table 2
Forward selection results for the two Redundancy Analyses (RDA) on soil microbial communities.

Name	Explains (%)	Contribution (%)	Pseudo-F	P	P (adj)
Interactive-forward-selection for bacterial communities (16S rDNA)					
Status_I	17.9	34.2	5.0	0.0002	0.00036
Status_T	7.6	14.5	2.2	0.0012	0.0027
Status_N-I	7.6	14.5	2.2	0.001	0.0027
C/N ratio	8.5	16.3	2.7	0.0002	0.0009
Interactive-forward-selection for fungal communities (18S rDNA)					
Status_N-I	13.2	27.8	3.5	0.0002	0.0006
Status_T	8.2	17.2	2.3	0.0002	0.0006
Status_I	8.2	17.2	2.3	0.0002	0.0006
C/N ratio	6.8	14.3	2.0	0.0034	0.00648
NO3	5.6	11.8	1.7	0.003	0.0063

P = P-values based on permutation test; P (adj) = P-values adjusted for false discovery rate following Benjamini and Hochberg (1995); Status_I = invaded; Status_T = transitional; Status_N-I = non-invaded.

results are consistent with other studies on invasive N_2-fixing tree species (*A. dealbata*, May and Attiwill, 2003; Lorenzo et al., 2010b; González-Muñoz et al., 2012; *Acacia* spp., Witkowski, 1991; Stock et al., 1995; *Robinia pseudoacacia*, Rice et al., 2004; *Myrica faya*, Vitousek and Walker, 1989; *Alnus* spp., Vogel and Gower, 1998). Many authors have also recorded changes in quantity or quality of litter composition registering a higher N content and a lower C/N and lignin-N ratios in leaf litter of invasive N_2-fixing stands than in litter from the dominant native tree (Corbin and D'Antonio, 2004; Marchante et al., 2008). Moreover, frequent invaders have faster growth rates than native species (Ehrenfeld, 2003) and the massive detritus production and differences in litterfall under *A. dealbata* canopy are possible explanations for the greater accumulation of soil N and C. Soil derived from transition patches showed intermediate values of C and N contents, between those of invaded and non-invaded areas, strengthening the hypothesis that alteration of soil environment is a result of *A. dealbata* invasion.

Soil acidification was frequently reported under leguminous species, as either direct or indirect effects of N_2 fixation, such as nitrification, NO_3 leaching and general imbalances in the C and N cycles (Tang et al., 1999). In our study, acidification caused by *A. dealbata* changed soil pH status from moderately to strongly acidic, leading to potential negative effects such as a decrease in phosphorus, calcium and magnesium availability, as well as aluminium toxicity.

As highlighted by RDAs on both soil bacterial and fungal communities, a direct effect of invading plant on microbial communities' composition can be hypothesized, as already shown by Si et al. (2013) on fungal communities. Novel chemicals released by *A. dealbata* through root exudates or leaf leachates may influence the catabolic diversity of resident microbial communities by changing the abundance of particular members (Van der Putten and van der Putten, 2010). Moreover, allelopathic compounds released by *A. dealbata* were found to be responsible for changes in the soil microbial communities (Lorenzo et al., 2013). Invasion of symbiotic N_2-fixing species into ecosystems in which such species are uncommon or absent may also affect the abundance and activity of no-symbiotic N_2-fixing bacteria (Ehrenfeld, 2003).

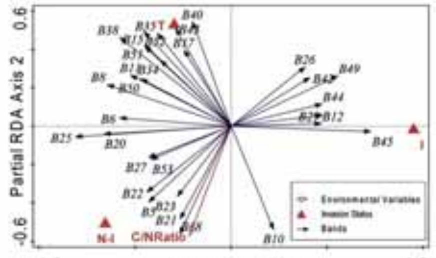

Fig. 2. Ordination diagram on soil bacterial communities based on 16S rDNA bands distribution obtained with partial-RDA analysis constrained by "invasion status" and C/N ratio according to the forward selection of explanatory variables (total variance explained by constrained axes 34.0%). 30 best fitting bands are shown. Invasion status: I = invaded; T = transitional; N-I = non-invaded.

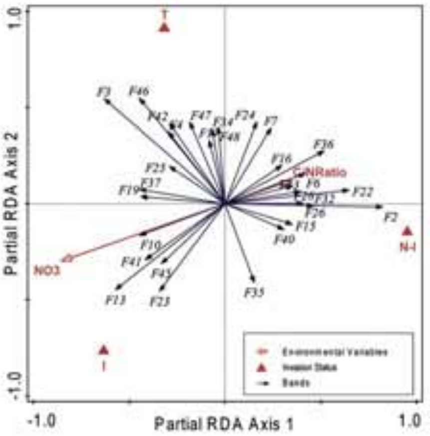

Fig. 3. Ordination diagram on fungal bacterial communities based on 18S rDNA bands distribution obtained with partial-RDA analysis constrained by "invasion status", C/N ratio and NO_3 content of soils according to the forward selection of explanatory variables (total variance explained by constrained axes 33.7%). 30 best fitting bands are shown. Invasion status: I = invaded; T = transitional; N-I = non-invaded.

A strong effect was also exerted on plant communities. Richness, diversity and total plant cover all showed lower values in the invaded than in non-invaded subplots, with transition generally in the middle. These results are consistent to other similar cases of study (Fuentes-Ramirez and Pauchard, 2010; González-Muñoz et al., 2012; Lorenzo et al., 2012), where *A. dealbata* was found to cause a decrease in total cover (Lorenzo et al., 2012), species richness and diversity. The multiple impacts of this invasive species appear to be related to different causes.

The structure of native vegetation can be deeply modified after *A. dealbata* invasion (González-Muñoz et al., 2012), determining variation in the photosynthetically active radiation (PAR) arriving on the ground (Lorenzo et al., 2012), leading to enhanced light and space competition (Jäger et al., 2009), thus determining an effect on the understory plant community. *A. dealbata* invasion in Elba Island strongly altered the structure of the native shrubland. An important role could also be played by allelopathic compounds and leaf leachates released by *A. dealbata*, although their effect could depend on soil conditions. *A. dealbata* leachates where firstly proved to inhibit seed germination and seedling emergence of the native plants and to increase the germination rate of its own seeds (Lorenzo et al., 2010c,2010d) but subsequently they were proved to exert a detrimental effect on the germination and growth of *A. dealbata* plants on native soils, elucidating that the effects of leachates depends on the origin of soils (Lorenzo and Rodríguez-Echeverría, 2012). Moreover, according to the findings of González-Muñoz et al. (2012), *A. dealbata* has a strong effect also on the seed bank.

Table 3
Results for the GLMMs on understory plant communities. Means (n = 9) calculated by invasion status ± standard errors are shown. Different letters in a column indicate significant differences at P < 0.05 between means according to the Post-hoc test.

	Invaded	Transition	Non-invaded	n DF	d DF	F-value	P-value
Species richness	3.33 ± 0.23 a	6.38 ± 0.45 b	7.04 ± 0.44 b	2	64	33.6633	<.0001
Diversity (H′)	0.9 ± 0.07 a	1.49 ± 0.09 b	1.39 ± 0.09 b	2	64	12.34541	<.0001
Total plant cover	44.67 ± 6.44 a	69.83 ± 5.44 b	80.42 ± 4.59 b	2	64	12.76948	<.0001
Percentage of nitrophilous species	0.29 ± 0.05 a	0.21 ± 0.03 b	0.09 ± 0.02 c	2	64	5.93662	0.0043
Relative abundance of nitrophilous species	0.27 ± 0.06 a	0.18 ± 0.04 b	0.04 ± 0.01 c	2	64	7.49534	0.0012

Finally, an important plant–soil feedback can be hypothesized, with *A. dealbata* taking advantage of the modification that the species determine on soil's biotic and abiotic compartment. Indeed, *A. dealbata* was found to have better performance in invaded than in non-invaded soils, mainly due to the modification of soil microbial communities (Rodríguez-Echeverría et al., 2013).

On the other hand, a series of indirect links could also be hypothesized. Much of the alteration to soil environment may be due to the altered microbial communities that subsequently modify nutrient pool and fluxes (Ehrenfeld, 2003; Jordan et al., 2008; Kao-Kniffin and Balser, 2008). According to Lorenzo et al. (2013), soil microbial communities may play an active role in the changes in soil nutrient fluxes. Conversely, soil chemical properties are expected to exerted strong effects on both soil microbial and aboveground understory plant communities.

UPGMA results supported the hypothesis that changes induced by wattle invasion on soil nutrient availability play an important role in shaping the composition of both bacterial and fungal communities. RDA analysis provides further evidences, showing that a number of microbial "sequence type" (phylotype according to Muyzer et al., 1995) are more abundant or even exclusive within a particular invasion status (invaded, non-invaded or transitional), whereas a number of species resulted equally distributed across the samples (Table S-3, Supplemental materials). Soil harbors a massive richness of microbial species (Torsvik and Øvreås, 2002) and soil type, pH and nutrient status are the main drivers controlling composition and diversity of soil microbial communities (Girvan et al., 2003; Fierer and Jackson, 2006; Pastorelli et al., 2013). Several recent works demonstrated that the soil microbial community is composed by a relatively small number of predominant taxa while a plethora of microbial species is present in low number, the so called "rare biosphere" (Elshahed et al., 2008). The dominant microbial groups play an active role in soil functions (Felske et al., 1997), while the rare species ("seeds") remain in a reversible state of dormancy or reduced metabolic activity fain to be trigged into activity by any changes in soil nutritional status and environmental conditions (Epstein, 2009; Lennon and Jones, 2011). This might explain that no significant or only marginal variations were observed for richness and diversity index of fungal and bacterial community, respectively.

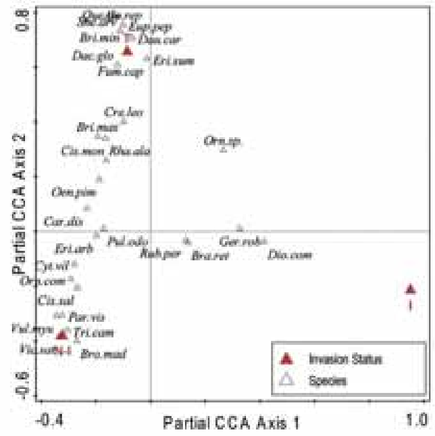

Fig. 4. Plant species distribution obtained with CCA analysis using "invasion status" as an explanatory variable (total variance explained by two constrained axes 8.1%; Pseudo F = 2.8, P = 0.0002). The 30 most fitting species are shown. Invasion status: I = invaded; T = transitional; N-I = non-invaded.

Moreover, RDA results highlighted the strong influence of C/N ratio and NO_3 soil content on the distribution through the invasion gradient of bacterial and fungal species.

Ge et al. (2010) indicated that the C/N ratio of decomposable substrates can be a major driver for the shift in the soil bacterial community. Although supported by marginal differences, our results agree with these findings as we found that bacterial richness and diversity were higher at higher C/N ratio (non-invaded area) than at lower C/N ratio (invaded area). These results may reflect a selection mechanism linked to organic matter quality and substrate decomposability, which may have favored microbial species more adapted to utilize available substrates, with a shift in microbial composition communities and a consequent decline in bacterial diversity.

Besides C/N ratio, fungal composition was significantly affected by NO_3 availability, according to the results of Balser and Firestone (2005), which found NO_3, more than NH_4, to be dominantly related to microbial community composition. A selection towards fungal species involved in nitrification processes (Schimel et al., 1984; Hart et al., 1997) or NO_3 assimilation (Merrick and Edwards, 1995; Marzluf, 1997) may be hypothesized, without changes either in fungal diversity or in richness.

The increase in N content is reflected also in the changes in species composition and in the decrease in richness and diversity of the plant taxa in invaded subplots, due to the increased competition. As highlighted by CCA results, plant community composition severely changed along the invasion gradient. The shift in species composition found in our stands appears to be mainly driven by the higher contribution of the guild of nitrophilous plants in terms of both relative species number and abundance in the invaded stands. This is a typical scenario for a "transformer", as have been already highlighted for *R. pseudoacacia* for both vascular plant (Benesperi et al., 2012) and epiphytic lichen communities (Nascimbene et al., 2012). The linkage between nitrophilous species and soil N content is also reflected in the way they change across the invasion status degree. In fact, both significantly increase from non-invaded to invaded areas, highlighting that the process of nitrification is operating already at the transitional invasion status. This should not be surprising where such changes are operating in a relative nutrient poor environment as the native matorral.

The link between plant communities and soil microbial communities appears unclear. It has been hypothesized that the greater plant richness found in non-invaded vegetation should lead to a greater soil biotic richness (Belnap et al., 2005) through the positive interaction involving litter quality and rhizosphere effect (Bohlen, 2006). Whereas a higher plant richness was found to be related to a higher richness of several groups in soil microbial communities in non-invaded compared to invaded stands (Belnap et al., 2005 on sites invaded by *Bromus tectorum*), in our case, this relationship appeared to be marginally true only for richness and diversity of bacterial communities (both higher in non-invaded soil samples, even if only marginally), while it was not found for fungal communities. Thus, it is not completely clear how plant richness on invaded areas can influence soil community performances in terms of richness and diversity and which kind of feedback processes could occur. Moreover, due to high redundancy of microbial function in soil, it has not been demonstrated that changes in specific soil microbial groups undermine the nutrient cycling and the sustainability of the soil ecosystems itself. On the other hand, Rodríguez-Echeverría et al. (2013) hypothesized that changes in soil microbiota induced by *A. dealbata* could determine a negative effect on the growth and performance of native species.

According to our results, *A. dealbata* highly impacted both the abiotic and biotic compartments of the local ecosystem and its ecological processes. Fig. 5 proposes a functioning framework hypothesizing the main relationships between the compartments we took into consideration, and representing the direct and indirect linkages with *A. dealbata* invasion in the specific situation we analyzed. Impacts vary across the invasion gradient; however, a low degree of invasion induces further changes in the soil properties and in the composition of vascular plant communities. Additional studies are needed towards a complete understanding of the processes involved in these changes, especially on the role of microbial communities in the modification of the soil properties and on their relationship with plant communities. However, while often the impacts related to alien plants are considered to be strongly contest-dependent and unpredictable (Hulme et al., 2013), our results are consistent with many other cases of study in both direction and magnitude of the recorded impacts, thus confirming the effectiveness of *A. dealbata* as invader and transformer across a wide variety of habitats and regions of the world.

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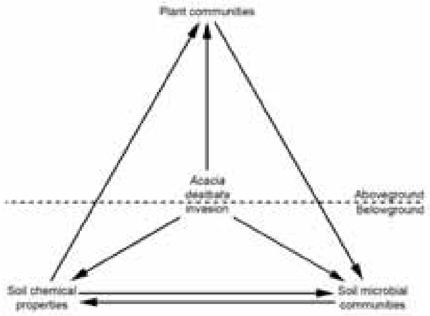

Fig. 5. Proposed functioning framework for *Acacia dealbata* impacts on the analyzed compartments of the local ecosystem. *A. dealbata* invasion directly impacts the quantitative and qualitative properties of soils, soil microbial communities and plant communities. In addition these compartments appear related and indirect effects due to invasion can be found between the different compartments.

Appendix A. Supplementary data

Supplementary data to this article can be found online at http://dx.doi.org/10.1016/j.scitotenv.2014.08.014.

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4.3 Plant species loss and community nestedness after leguminous tree Acacia pycnantha invasion in a Mediterranean ecosystem

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ABSTRACT

Invasion of ecosystems by alien species results as one of the major threats to biodiversity. Among the alien plant species, the ones belonging to the family Fabaceae s.l. represent some of the most dangerous and well known global invaders. In the Mediterranean ecosystems many Australian acacias were found to establish and rapidly spread often determining severe impacts on the understory vegetation. In the present work, we report the case study of *Acacia pycnantha* invasion in a typical Mediterranean matorral (Elba Island, central Mediterranean Sea, Italy). We conducted a survey on understory plant communities across an invasion gradient from non-invaded to transitional and invaded areas, aiming to quantify the impacts on the understory assemblage and to investigate the ecological processes involved in the changes in species composition.

The understory plant community was highly affected also starting from the intermediate stage of invasion. Species richness, diversity and total cover were all lower in the invaded than in the non-invaded plots, with transition generally in the middle. In addition, plant community composition severely changed along the invasion gradient. The species set we recorded showed a nested structure, with the composition of species in the invaded plots representing a subset of the others. According to our findings *A. pycnantha* exerted detrimental impacts on the native vegetation mainly determining a severe species loss in the understory assemblage and leading to an impoverishment of the invaded ecosystems.

KEYWORDS

Australian acacias; native understory plants; species richness; impacts on ecosystem; ecological processes

INTRODUCTION

Impacts related to invasive alien plants have been recognised as one of the major threats to biodiversity (CBD decision VI/23 2014). Invasive alien plants can affect plant community structure and composition (Daehler and Strong 1994; Hejda and Pyšek 2006; Gaertner et al. 2009; Nascimbene and Marini 2010) leading to an alteration of soils and vegetation properties (Vilà et al. 2006). They can cause a decrease in native plant and animal species richness and diversity (Pyšek et al. 2012; Vilà et al, 2014), especially at a local scale (Vilà et al. 2011), leading to local species loss (Powell et al. 2011). Impacts of invasive alien plant can be also reflected in changes in soil microbial communities (Marchante et al. 2008), and ecosystem processes such as nutrient and water cycling (Vitousek et al. 1987; Ehrenfeld 2003; Yelenik et al. 2007; Marchante et al. 2008). Impacts on species and ecosystems include also genetic variation via hybridization with native populations (Vilà 2000). Alien species can also lead to the *biotic homogenization* of invaded ecosystem, defined as the increase in biological similarity between communities through time (Mc-Kinney & Lockwood 1999, Olden & Poff 2004). Moreover nitrogen-fixing species could determine the facilitation of nitrophilous species due to nitrogen enrichment in soils (Le Maitre et al. 2011) and leading to a shift in species composition of invaded ecosystem towards more nitrophilous assemblages [i.e. *Robinia pseudacacia* for both vascular plant (Benesperi et al. 2012) and epiphytic lichen communities (Nascimbene et al. 2012)].

Australian acacias (Fabaceae, Mimosoideae) are a group of globally introduced species including at least 23 well known or emerging invaders in many parts of the world, especially in Mediterranean ecosystems, leading to a wide range of ecological and socio-economic impacts (Lorenzo et al. 2010; Le Maitre et al. 2011; Fuentes-Ramírez and Pauchard 2010; González-Muñoz et al. 2012; Lorenzo et al. 2012). Originary of Southern Australia and elected as national plant (Carruthers et al. 2011), *Acacia pycnantha* is nowadays distributed out of its native range in Europe and South Africa (Richardson and Rejmánek 2011). Gassó et al. (2010) and Wilson et al. (2011) reported this species as potentially invasive and it is often mentioned as a problematic invader for South Africa (Hoffmann et al. 2002; Nel et al. 2004; Carruthers et al. 2011). Although being listed as casual in Italy (Celesti-Grapow et al. 2009), the species was recently reported as naturalized in Sardinia (Podda et al. 2012) and as invasive in Tuscany (Lazzaro et al. 2014a).

Although this species is a widely known invader, no quantitative information can be retrieved on its impacts. Indeed studies on impacts by invasive Australian acacias are mainly focused on a relatively small group of deeply investigated species (e.g. *Acacia dealbata*, *A. longifolia*). In addition, investigations on the invasion by less studied species are urgently demanded (Hulme et al., 2013), representing a valuable source of information to lay the basis for any generalization on the scenario of biological invasions.

Aiming to produce a valuable contribution to the knowledge of the effects of this poorly studied species, we analysed the presence of impacts by *A. pycnantha* invasion in the Elba Island (Central Italy), specifically evaluating species richness, diversity and composition of the understory vegetation. We focused on the following main questions: (*i*) Does *A. pycnantha* qualitatively and quantitatively impact the understory plant communities? (*ii*) Which ecological processes are involved in the alteration in species composition? Towards these aims, we conducted a sampling of understory plant communities across a gradient of invasion going from invaded to transitional and non-invaded vegetation.

METHODS

Study area

The present study was carried out in the South-East of Elba Island, the largest of the seven main islands of the Tuscan Archipelago, Central Mediterranean Sea (Tuscan Archipelago National Park, Central Italy). The study area includes an area of about 1 square kilometer, with elevation ranging from 250 m up to 300 m above sea level invaded by *A. pycnantha*. Native vegetation in the study area is a typical Mediterranean vegetation constituted by a matorral dominated by *Erica arborea* and *Calicotome villosa*, representing the degradation of the native *Quercus ilex* forest. *A. pycnantha* was introduced in the area, with *A. dealbata*, in the second half of 20th century generally mixed with *Pinus* spp., *Quercus ilex* and *Fraxinus ornus* (Gatteschi and Arretini 1990). In 1998 a fire completely displaced the pine plantation determining the spread of *A. pycnantha*. Nowadays several pure invaded stands ranging from about 1000 square meters up to 5 hectares of surface are present in the area of study. The study area is homogeneous for the climate (thermomediterranean climate: Foggi et al. 2006) and for the geological substratum (classified as Eutric Cambisols developed on parental material mainly composed of metamorphic rock: white schists; Costantini et al. 2012).

Sampling design

The sampling of plant communities took part in May 2013, adopting a sampling method with a hierarchical structure. We selected three patches invaded by *A. pycnantha* (macroplots) for the vegetation survey. In each macroplot we defined three levels of invasion (hereafter referred as “invasion status”) according to the gradient of invasion of the native communities: (1) ‘invaded vegetation’, dominated by *A. pycnantha*, (2) ‘non-invaded vegetation’, consisting of contiguous mature communities with a predominant native understory without any *A. pycnantha* individuals, and (3) ‘transition vegetation’, between the invaded and non-invaded vegetation, consisting of both small wattle trees and native shrubs.

Along the invasion gradient we performed a sampling of the understory plant communities. For each macroplot we randomly selected two transects 2 m wide (at least 20 m apart), going from the invaded, through the transitional up to the non-invaded vegetation. Within each transect we sampled the understory vegetation in four 0.5 m side quadrate plots randomly selected for each invasion status, including only herbs, and immature shrubs and trees lower than 30 cm. In each quadrat we recorded the species abundance as the number of individuals for each species (for the species *Brachypodium distachyon*, *B. ramosum* and *Bromus madritensis* the number of individuals were only estimated when greater than 25, due to difficulties in determining the real number in this case). We also recorded the total plant cover (%), estimated as the area covered by the all vascular plant species in the quadrate, determined by dividing the quadrate into 100 5 x 5 cm cells and adding the number of occupied cells. A total of 72 plots (3 macroplots x 2 transects x 3 invasion status x 4 quadrate plots) were sampled.

Data analysis

We analyzed plots species richness, species diversity, expressed by Shannon index H’ (Shannon and Weaver 1949), and total plant cover aiming to investigate how understory plant communities were affected by *A. pycnantha* invasion.

Given the hierarchical structure of the data we performed the analyses adopting the framework of Generalized Linear Mixed Models (GLMMs) to get rid of the variation deriving from the macroplot and the transect of origin of the plots. We run the GLMMs adopting Restricted Maximum Likelihood Method (REML). A logarithmic scale transformation was adopted in the case of species richness, while an arcsine transformation was used for total plant cover data (aiming to normalize the residuals). We used the “invasion status” of the plots as explanatory variable with fixed effect. The variables "Macroplot" and "Transect" were used as random effect factors, considering the variable "Transect" nested in "Macroplot". If the main factor was significant, we used a Tukey test to investigate the differences between the three levels of invasion.

To test how the invasion status was affecting the composition of the plots we performed a partial-Canonical Correspondences Analysis (CCA) using the species abundances. The variable "Transect" was used as a covariate to get rid of partial variation due to this grouping factor and the "Invasion Status" of the plots was used as explanatory variable. The significance of all the CCA constrained axes was tested with a permutation test based on the hierarchical structure of the data (with 4999 permutations), with freely exchangeable permutations run inside each transect. A logarithmic transformation was applied to the species abundances and while the result of this kind of analysis is usually affected by the presence of rare species these were down-weighted in the computations. Particularly, following default option in Canoco 5, rare species are those whose relative frequency of nonzero values is below 20% of the frequency of the most frequent one, in this case the relative weight of the species was decreased proportionally to their frequency.

Finally, we studied the degree of nestedness of the dataset aiming to highlight the fact that composition of invaded plot represents a subset of non-invaded and transitional ones. According to this aim, we used the nestedness metric based on overlap and decreasing fill (NODF, Almeida-Neto et al. 2008). NODF allows to combine two independent nestedness metrics which quantify (1) whether depauperate assemblages constitute subsets of progressively richer ones (NODFsite) and (2) whether less frequent species are found in subsets of the sites where the most widespread occur (NODFspecies) (Ulrich et al. 2009). According to our aim, we partial NODF statistics with different meanings. Particularly we studied the NODFsite statistic, which express the degree to which species poorer sites form compositional subset of species richer sites. We evaluated the statistical significance of nestedness in our dataset comparing the results with a fixed-fixed null model as recommended by Ulrich et al. (2009). Thus our dataset is compared to null model which maintains both the number of species in the plots and the frequencies of the single species.

The GLMMs were carried out using the *nlme* package (Pinheiro et al. 2014), and the postHOC tests were carried out exploiting the *multcomp* package (Hothorn et al. 2013) for R software version 3.1.0 (R Core Team 2014). The nestedness analyses (NODFsite and null model comparison) were performed using the *vegan* package vers. 2.0-10 (Oksanen et al. 2013) for R software. The ordination analysis were performed using Canoco 5 for Windows (vers. 5.03, Ter Braak and Šmilauer 2012)

RESULTS

We detected a total of 57 plant species (19 in the invaded, 36 in the transitional and 44 in the non-invaded plots). 11 of these were common to all the three statuses of invasion, whereas 5 were exclusive of the invaded quadrates, 7 of the transition ones and 14 of the non-invaded. 1 species was detected in both the invaded and the transitional subplots, 17 in both the transitional and the non-invaded subplots, and 2 species were shared exclusively between the invaded and the non-invaded plots (Appendix 1).

According to GLMMs results we found significant differences for all the variables analyzed (Table 1). Species richness decreased significantly going from the non-invaded throughout the transitional, up to the invaded plots (Fig. 1a). Diversity was higher in the non-invaded plots than in the invaded ones with the transitional plots being similar to the non-invaded ones (Fig. 1b). Total plant cover was higher in the non-invaded plots than in the invaded ones (Fig. 1c).

model		**numDF**	**denDF**	**F-value**	**p-value**
Species Richness	Invasion status	2	64	49.61149	<.0001
Diversity (H')	Invasion status	2	64	21.77655	<.0001
Total Plant Cover	Invasion status	2	64	35.43509	<.0001

Table 1 Anova table for GLMMs performed to compare plot level species richness, Diversity (H') and Total Plant Cover by invasion status. numDF = numerator degree of freedom; denDF = denominator (residual) degree of freedom.

According to the global permutation test on constrained axes of the partial CCA the invasion status significantly affected the plot species composition (Pseudo F =2.5, P=0.0002). The constrained axes of CCA (Fig. 2) accounted for 7.3% of the partial variation (explained variation axis 1 = 4.44%; axis 2 = 2.86%, partial variation = 5.7). According to these results, invasion status appeared lying on the first axis, as confirmed by the intermediate position of the transitional status in comparison to the non-invaded and invaded statuses. Furthermore, as the

CCA is based on double data standardization, it is noteworthy that relative species proportions are different under different invasion statuses.

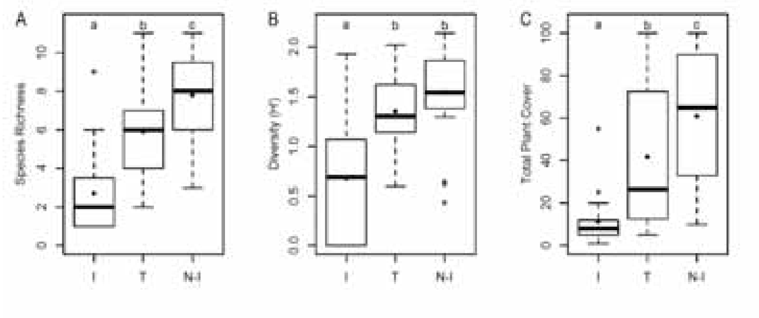

Fig. 1 Box plot diagrams for Species Richness (A), Shannon Diversity H' (B) and Total Plant Cover (C) of plots grouped by invasion status. Bold line: median, rhombus: mean, box: interquartile range, whiskers: typical range, points: outliers. Different letters indicate significance differences between means at P = 0.05 level according to the PostHOC test. (Invasion Status: I =Invaded; T= Transitional; N-I = Non-invaded)

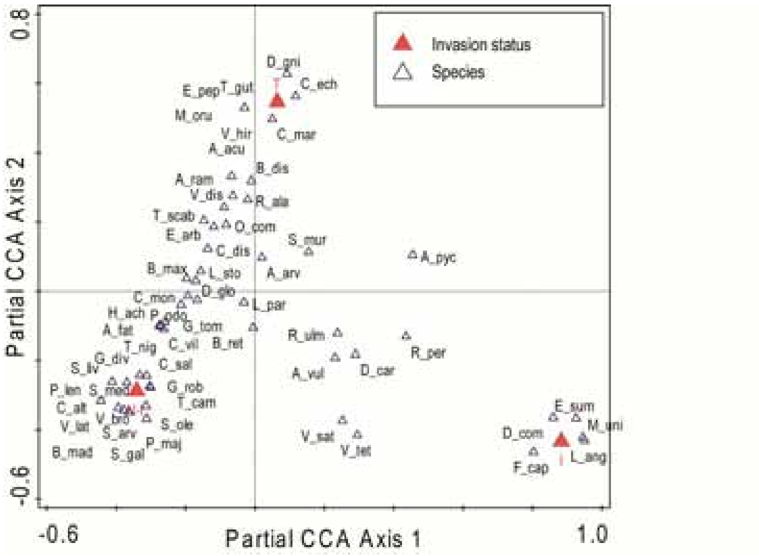

Fig. 2 Plant species distribution obtained with partial-CCA analysis using “invasion status” as a explanatory variable (Total variance explained by two constrained axes 7.3%, axis 1 = 4.44%, axis 2 = 2.86%, partial variation = 5.74626; Pseudo F = 2.5, P = 0.0002). Invasion Status: I =Invaded; T= Transitional; N-I = Non-invaded. Species short names are jittered to facilitate the reading of the figure. Variable "invasion status" appear lying on the firs axis, with transitional invasion status between non-invaded and invaded.

Composition of species in the plot compared to null model had a significantly higher degree of nestedness than 95% of null models (NODFsite = 27.756; Pvalue = 0.027 after 999 simulations). Moreover the plots were automatically ordered with the non-invaded and transitional plots on the top of the matrix and the invaded ones on the bottom, according to the number of species they hosted, and highlighting the fact that the invaded plots represent a subset of the others (Fig. 3).

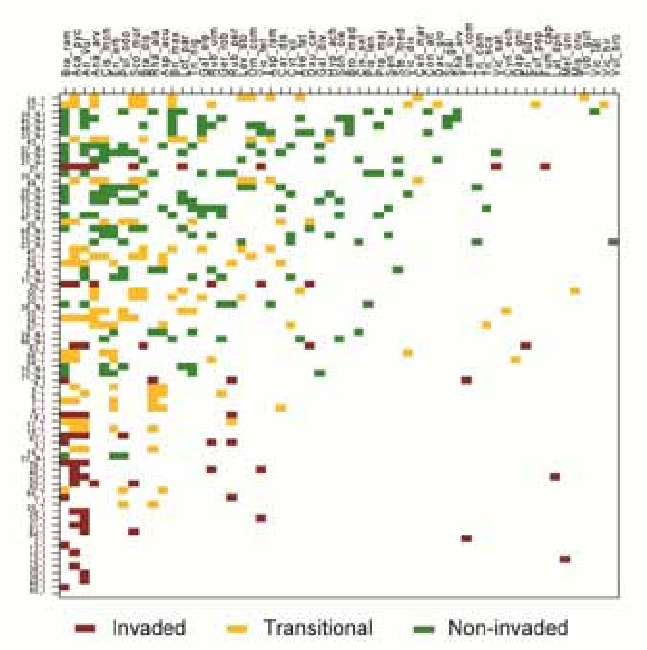

Fig. 3 Nestedness analysis plot: Species–site matrix for 57 species (columns) recorded in 72 plots (rows). Species presences are marked with a full square. Plots (indicated by numbers 1–72 and Invasion Status: I =Invaded; T= Transitional; N-I = Non-invaded.) are ranked by species richness and species are ranked by frequency. The squares marking the species presences are coloured according to the invasion status of the referring plot: red for Invaded plots, orange for Transitional ones and green for Non-invaded ones

DISCUSSION

The aim of the study was to assess the effects of *A. pycnantha* invasion on species richness and composition of the understory plant communities. According to our findings, the understory plant community was highly affected also starting from the intermediate stage of invasion. Species richness, diversity and total cover were all lower in the invaded than in the non-invaded plots, with transition generally in the middle. Moreover plant community composition severely changed along the invasion gradient.

In South African fynbos, where *A. pycnantha* is reported as invasive (van Wilgen et al. 2011), invasion by Australian acacias is leading to a change in community structure, to an alteration of both nutrient and water cycling, and to a reduction in native plant richness (Richardson and Kluge 2008). Unfortunately, given the lack of quantitative information of the impacts exerted by this species, especially in Europe and in the Mediterranean region, it was not possible to compare our data to other cases of study. However, our findings on the impacts of *A. pycnantha* on the native plant communities are consistent with other cases of invasion by leguminous trees such as the congeneric *A. dealbata*. This species led to a decrease in species richness, diversity and in total plant cover of the understory vegetation and to a strong influence in the species composition of the communities (González-Muñoz et al. 2012; Lorenzo et al. 2012; Lazzaro et al., 2014b). This is also consistent with the findings of Vilà et al. (2014), which highlighted that closely related non-native species exert similar impacts on native communities. The severe impacts of acacias are related to some key traits such as the high growth rate and biomass accumulation, the production of a large and persistent seed bank, the capacity to establish associations with nitrogen-fixing bacteria (Le Maitre et al. 2011), and the release of allelopathic compounds (Lorenzo et al. 2010).

In our case CCA results underlined changes in the plot species composition at different invasion statuses. Furthermore, many species were associated to the non-invaded and the transitional plots whereas few species are associated to the invaded ones. Thus the main driver of differentiation appeared to be the decline in species number going from the native to the invaded vegetation. Moreover, the nestedness analysis highlighted that the invaded plot species composition represented mostly a subset of the non-invaded and the transitional plots, with very few species entering in the invaded sites to replace the lost ones. Therefore the exclusion of most of the species appeared to be the main ecological process acting already at the transitional stages and completely developed at the invaded stages. This process is probably mainly driven by the nitrification process and the subsequent changes in the litter layer. *Acacia*'s species as nitrogen-fixing plants are known to greatly improve soil nitrogen concentration (Yelenik et al. 2004; Lorenzo et al. 2010), thus influencing plant species composition and richness. The increase in nitrogen supply is recognized as one of the main threats to natural vegetation (Hicks et al. 2011) and is likely to strongly influence species richness and species assemblage of understory plant communities. Species loss is frequently reported following nitrification of soils (Clark et al. 2007), mainly due to resource-based competition (Honsová et al. 2007; Kirkham et al. 2008). On the other hand, as found for other invasive alien legume species (*Robinia pseudoacacia*, Benesperi et al. 2012; Nascimbene et al. 2012), invasion by nitrogen-fixing trees could determine a shift in the species composition in favor of nitrophilous species. Nevertheless in our case such trend was not recorded, as also highlighted by the high number of common species between the invaded and the transitional or the non-invaded plots. Actually only 5 species are exclusive of the invaded stands, thus representing a low level of replacement.

The high amount of litter accumulation could play a determinant role in the species selection, acting as a source of organic nitrogen and also leading to accumulation of allelochemical compounds in the soils. Indeed, phytotoxic and allelopathic capacities have been reported for many wattle species (Lorenzo et al. 2010).

Many of the species vanished from the invaded status are represented by the saplings of typical Mediterranean shrubs, such as *Erica arborea*, *Cistus monspeliensis*, *Cytisus villosus*, *Cistus salviifolius* and *Pistacia lentiscus*, whose abundance gradually decrease from the non-invaded to the transitional plots. This trend underlines the changes in the native vegetation structure, and the loss of renovation of the typical dominant species. Furthermore, the gradual disappearance of many typical Mediterranean herbaceous species, (such as *Bromus madritensis*, *Hypochaeris achyrophorus*, *Senecio lividus* and *Galium divaricatum*) witnesses the impoverishment in the understory herbaceous strata, finally leading to the low-cover understory community that characterize the invaded sites.

In addition, it is noteworthy the presence of an alien species as *Erigeron sumatrensis*, which was only found in a single invaded plot, underlining the risk of “invasional meltdown process” (Simberloff and Holle 1999), further enhanced by the high invisibility of invaded and impoverished sites. *A. pycnantha* may play an essential role in the permeability of invaded habitats to the ingression of other alien species, such as *E. sumatrensis*, simplifying the ecological community and lowering the resilience of the ecosystem.

Our results document the strong impacts related to the invasion by *A. pycnantha*, highlighting the presence of severe species loss in the transitional and the invaded sites, with invaded understory communities being a impoverished subset of the native ones. Changes are gradual and the impoverishment of species from the native communities started already in the transition plots, causing a lack of renovation of the native shrubland and to the complete displacement of native communities with a new one, low diverse and poor, largely dominated by the invasive species. These results highlights that immediate action is needed to protect and maintain the diversity of native plant communities. Further studies are needed to better understand the ecological processes acting during the invasion and the potential effect of litter layer and allelopathy related to this invasive species.

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Appendix 1 Plant species sampled in the three invasion statuses ordered alphabetically. Total species occurrences and relative abundance per Invasion Status expressed as percentage of the total individuals found are shown. The abbreviation adopted in the CCA diagram is given. Nomenclature follows the online databases Euro+Med (2006-2014) (Euro+Med PlantBase - the information resource for Euro-Mediterranean plant diversity: http://ww2.bgbm.org/EuroPlusMed/ [accessed 2014 May 7]) and The Plant List. (version 1.1: http://www.theplantlist.org/. [accessed 2014 May 7]).

Species	Abbreviation adopted in the CCA diagram	Species occurences	Invaded	Transition	Non-Invaded
Acacia pycnantha Benth.	A_pyc	29	45	55	0
Anagallis arvensis L.	A_arv	21	14	48	38
Arisarum vulgare O.Targ.Tozz.	A_vul	29	41	24	34
Asparagus acutifolius L.	A_acu	11	0	64	36
Asphodelus ramosus L.	A_ram	6	0	67	33
Avena fatua L.	A_fat	5	0	40	60
Brachypodium distachyon (L.) P.Beauv.	B_dis	13	8	62	31
Brachypodium retusum (Pers.) P.Beauv.	B_ret	36	28	25	47
Briza maxima L.	B_max	11	0	45	55
Bromus madritensis L.	B_mad	3	0	0	100

Species	Abbreviation adopted in the CCA diagram	Species occurences	Invaded	Transition	Non-Invaded
Carex distachya Desf.	C_dis	6	0	50	50
Centaurium maritimum (L.) Fritsch	C_mar	2	0	100	0
Cistus monspeliensis L.	C_mon	18	0	39	61
Cistus salviifolius L.	C_sal	3	0	0	100
Convolvulus althaeoides L.	C_alt	2	0	0	100
Cynosurus echinatus L.	C_ech	1	0	100	0
Cytisus villosus Pourr.	C_vil	6	0	17	83
Dactylis glomerata L.	D_glo	2	0	50	50
Daphne gnidium L.	D_gni	1	0	100	0
Daucus carota L.	D_car	5	40	20	40
Dioscorea communis (L.) Caddick & Wilkin	D_com	2	100	0	0
Erica arborea L.	E_arb	17	0	47	53
Erigeron sumatrensis Retz.	E_sum	1	100	0	0
Euphorbia peplus L.	E_pep	1	0	100	0
Fumaria capreolata L.	F_cap	1	100	0	0
Galactites tomentosa Moench	G_tom	4	0	0	100
Galium divaricatum Pourr. ex Lam.	G_div	9	0	33	67
Geranium robertianum L.	G_rob	8	0	13	88
Hypochaeris achyrophorus L.	H_ach	4	0	25	75
Lathyrus angulatus L.	L_ang	7	0	57	43
Lavandula stoechas L.	L_sto	1	100	0	0
Lotus parviflorus Desf.	L_par	10	10	30	60
Melica uniflora Retz.	M_uni	1	100	0	0
Misopates orontium (L.) Raf.	M_oru	1	0	100	0
Ornithopus compressus L.	O_com	7	0	57	43
Pistacia lentiscus L.	P_len	3	0	0	100
Prasium majus L.	P_maj	3	0	0	100
Pulicaria odora (L.) Rchb.	P_odo	16	6	31	63
Rhamnus alaternus L.	R_ala	12	8	58	33
Rubia peregrina L.	R_per	8	50	25	25
Rubus ulmifolius Schott	R_ulm	9	33	22	44
Scorpiurus muricatus L.	S_mur	15	20	53	27
Senecio lividus L.	S_liv	3	0	0	100
Sherardia arvensis L.	S_arv	2	0	0	100
Silene gallica L.	S_gal	2	0	0	100
Sonchus oleraceus (L.) L.	S_ole	4	0	0	100
Stellaria media (L.) Vill.	S_med	3	0	0	100
Trifolium campestre Schreb.	T_cam	2	0	0	100
Trifolium nigrescens Viv.	T_nig	10	0	20	80
Trifolium scabrum L.	T_sca	2	0	50	50
Tuberaria guttata (L.) Fourr.	T_gut	1	0	100	0
Vicia disperma DC.	V_dis	1	0	0	100
Vicia hirsuta (L.) Gray	V_hir	2	50	0	50
Vicia lathyroides L.	V_lat	3	0	67	33
Vicia sativa L.	V_sat	1	0	100	0
Vicia tetrasperma (L.) Schreb.	V_tet	7	57	0	43
Vulpia bromoides (L.) Gray	V_bro	1	0	0	100

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5. RISK OF INVASION

It is globally recognized that management opportunities in case of IAS are mainly restricted to the earlier phases of the invasion (McNeely 2001). The CBD proposes three successive steps in IAS management: prevention, eradication and, if neither of the other steps is possible, control (CBD 2014). Prevention, thus not allowing a potentially invasive species to be introduced or to become established in the first place, is the first line of defense and the more efficient management option in terms of costs/benefice. In fact, the rapid reproduction and spread of the invasive species over time lead to an exponential increases in both the total area infested and the associated control costs. Once an alien species has established the control costs increase, while the likelihood of success decreases, as much as the species become a widespread invasive, when the economic and often environmental costs of its eradication can be unbearable (McNeely 2001). In this case, local control and mitigation of the impacts are often the only management options (see fig 5.1)

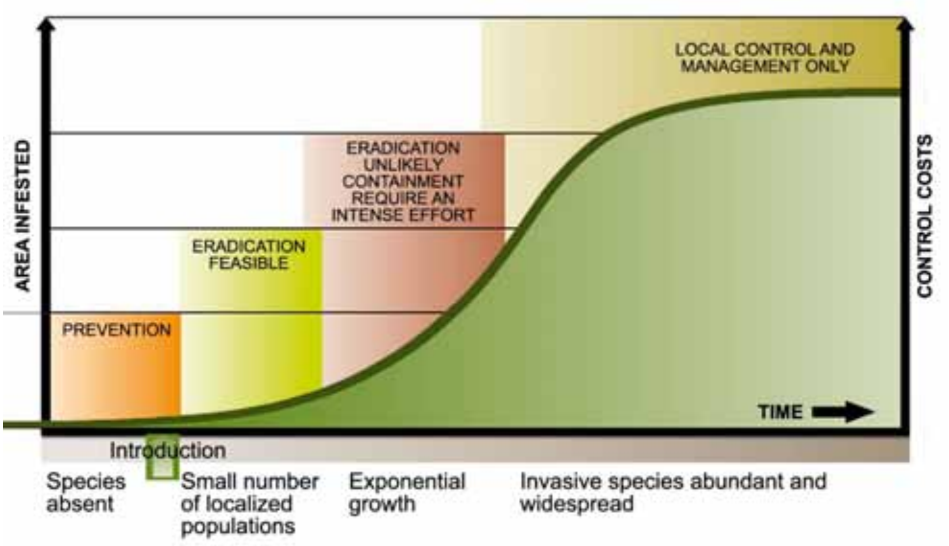

Fig 5.1 Invasion curve and management options across the different phases of the invasions. After introduction IASs pass a lag phase after which their spreading lead to exponential increases in both the total area infested and associated control costs.

While prevention is the most effective action for managing invasive species, early detection and rapid response methods are necessary to prevent infestations and control costs from reaching unmanageable levels. Therefore the early, ideally *ex ante*, identification of IAS is an urgent need (Essl et al. 2011). It is thus essential to develop and adopt early warning strategies allowing to predict potential new invasive species for a region or site, and/or predict potential new invasion sites for an invasive species (McNeely 2001). Particularly the identification of potential future IAS not yet present and the prioritization of alien species already present according to their impact (Gassò et al. 2009), or the individuation of those areas that are more prone to be colonized in order to optimize monitoring actions (Cronk & Fuller 1995) are essential in the management of biological invasion. These strategies benefit from the development and application of tools and procedures allowing to evaluate the risk of invasion, thus driving stakeholders and land managers in the prioritization of control efforts. The information on the traits of invasive species together with the characteristics of invaded habitats, and evidence of invasiveness in other parts of the world, especially in areas with similar environmental conditions, are the basis to build Risk Assessment (RA) procedures. RAs allow, driving the user across an exhaustive gathering of information on a certain species, to predict the success of this species in a given region of introduction. RAs usually focus on the evaluation of the two main components of the risk of invasion: the likelihood of invasion and the impacts, attempting to identify those species that are more likely to spread once introduced and consequently to produce impacts on the invaded ecosystems (Gassó et al. 2009 and references therein). This dual component of RAs is essential also in the individuation of the areas more prone to be invaded. In this case it is in fact important to focus on the areas where the potential invaders would affect valuable biota or habits worthy of conservation.

In this framework we aimed to evaluate the risk of invasion in the Tuscan Archipelago, facing the problem following two approaches aiming to 1) prioritize the alien species in TANP according to their invasiveness adopting and testing two Risk Assessment procedures; 2) identify the areas exposed to a greater risk of invasion, for the Island of Elba. These goals were again part of the TANP CoREM project, and aimed to produce valuable contributions towards the prioritizations of IAS management and control efforts by TANP.

Prioritization of the alien species introduced in TANP

To assess the risk of invasion across the alien plants already present in the TANP and prioritize the species according to their invasive potential we adopted two RAs, comparing their results and aiming to investigate the differences across the two schemes. We choose to test the European and

Mediterranean Plant Protection Organization (EPPO) Prioritization Process (EPP), and the Australian Weed Risk Assessment (A-WRA). EPP is a rapid screening prioritization proposed quite recently (Brunel et al. 2010; EPPO 2012), that could represent a unified and adaptable tool in the risk assessment of invasive alien species in Europe. On the other hand A-WRA is one of the first RAs developed (Pheloung 1999) and has been applied on large number of plants and in many countries.

EPP merge together information on the distribution and biogeography of the species, its spread potential and its capability to exert ecological or socio-economic impacts (Fig 5.2). According to this information the species are included in three main groups, with decreasing level of risk: the list of invasive (or potential in case they are still not present in the area under assessment) alien plants; the observation list and the minor concern list. Moreover, the uncertainty in the assessment, tracked for all the answers given during the assessment can be used, by means of Bayesian statistic, to rank the species according to overall uncertainty of the assessment.

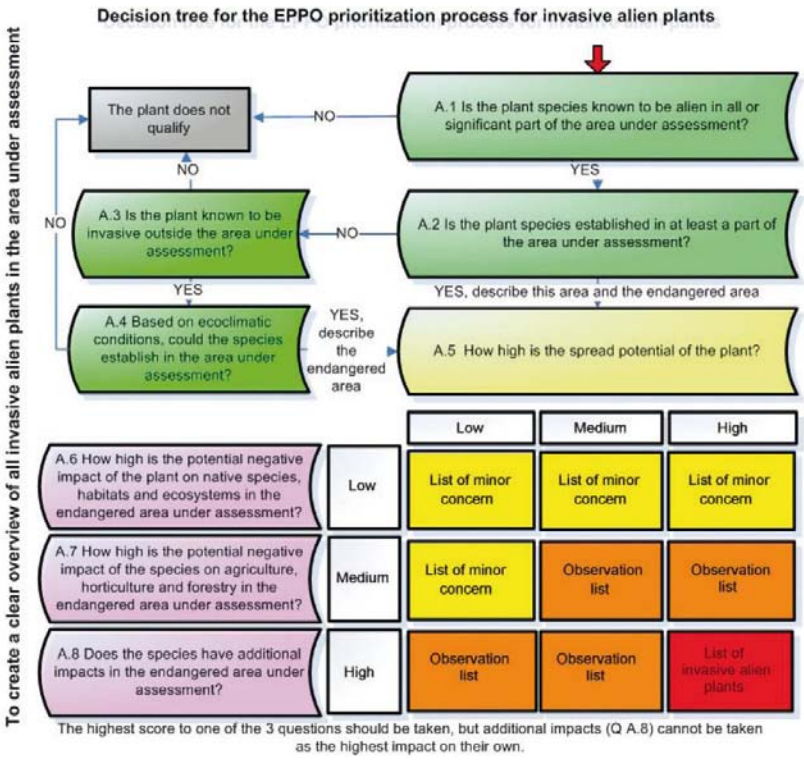

Fig 5.2 Decision tree for the EPPO prioritization process for invasive alien plant. From EPPO (2012).

The A-WRA is based on 49 questions, again regarding distributional and biogeographical traits of the species, undesirable traits (related to potential impacts exerted by the species) and to biology and ecology of the species (drawing the likelihood of the species to be a potential invader). Most of the responses are translated into a numerical score, positively correlated to weed potential. The invasiveness of the species is evaluated following a scoring system obtained summing the numerical scores of the answers. A high score (>6) identifies a species likely to be of high risk (weed) and rejects it for import; a low score (<1) accepts the plant for import (non-weed) and intermediate scores (1–6) require further evaluation.

To test the two methods, and to assess the risk of invasion by alien plat in TANP, we selected a large set of alien species (212 species), including all those casual or naturalized listed in the checklist of the alien flora of the Tuscan Archipelago, and also a wide set of species only-cultivated in the Tuscan Archipelago. Thus, for each of the species, we run both the EPP and the A-WRA, and compare the results of the two procedures. Moreover to evaluate the characteristics of the tested RAs we compared the results of both methods with the categorization done at national level by a panel of experts.

The results of the assessments and the comparison of the two RAs were preliminary presented as posters at international conferences. A first contribution was presented in 2012 at NEOBIOTA 2012 congress (NEOBIOTA 2012 - Halting Biological Invasions in Europe: from Data to Decisions; 7th European Conference on Biological Invasions; Pontevedra (Spain), 12-14 September 2012). Thus further progress in the research were presented in 2013 at the the EMAPI 2013 congress (12th Reunion on ecology and management of alien plant invasions; Pirenópolis, Goiás, Brazil; 22-26 September 2013) (See sections 5.1 and 5.2). Finally the work is object of a contribution (in prep. for Biological Invasions: Sec 5.3).

Prioritization of potential invasion sites for the most harmful species in the island of Elba

To identify the areas exposed to a greater risk of invasion in the Island of Elba we started from the concept to merge together the likelihood of invasion with the presence of areas worthy of conservation. The likelihood of invasion was assessed adopting Species Distribution Models (SDMs), while Natura2000 habitats were used as a proxy of valuable biota threaten by IAS. From the alien flora of TANP, we selected six particularly harmful species in Mediterranean Islands Ecosystems (Brundu et al. 2013). We choose to perform this analysis only for the Island of Elba, because only for this island we were able to collect the proper set of environmental and distributional data allowing a robust modelling process.

Fig 5.3 The six Invasive Alien Plants selected for the SDMs elaboration. (A) = *Acacia dealbata*, (B) = *Agave americana*, (C) = *Ailanthus altissima*, (D) = *Opuntia ficus-indica*, (E) = *Oxalis pes-caprae* and (F) = *Robinia pseudoacacia*.

The selected species are *Acacia dealbata*, *Agave americana*, *Ailanthus altissima*, *Opuntia ficus-indica*, *Oxalis pes-caprae* and *Robinia pseudoacacia*. The distributional information necessary for the modelling step was collected gathering all literature and observational information with a good spatial accuracy, merged from the data obtained with an exhaustive field survey aimed to map the distribution of these species in the Island of Elba.

We modelled the potential distribution of these species using the package *biomod2* (Thullier et al., 2014) for the statistical software R. This package implement the main state of the art modelling techniques, allowing to run several models for each species and to merge the best fitting models in an ensemble model. The advantage of this approach is to disregard the results from a specific mathematic algorithm, leading to results that are more general. Then the potential distribution of the six invasive species were merged together to obtain a distribution of threat of invasion, considered as the likelihood of invasion by the six worst invasive species of TANP.

The map of threat of invasion was merged with the map of density of Natura2000 habitat in the island of Elba, representing the valuable biota exposed to potential impacts, obtaining a map of the risk of invasion for the Elba Island.

The results of this procedure was submitted as a contribution to the journal *Environmental Conservation* (Sec 5.4).

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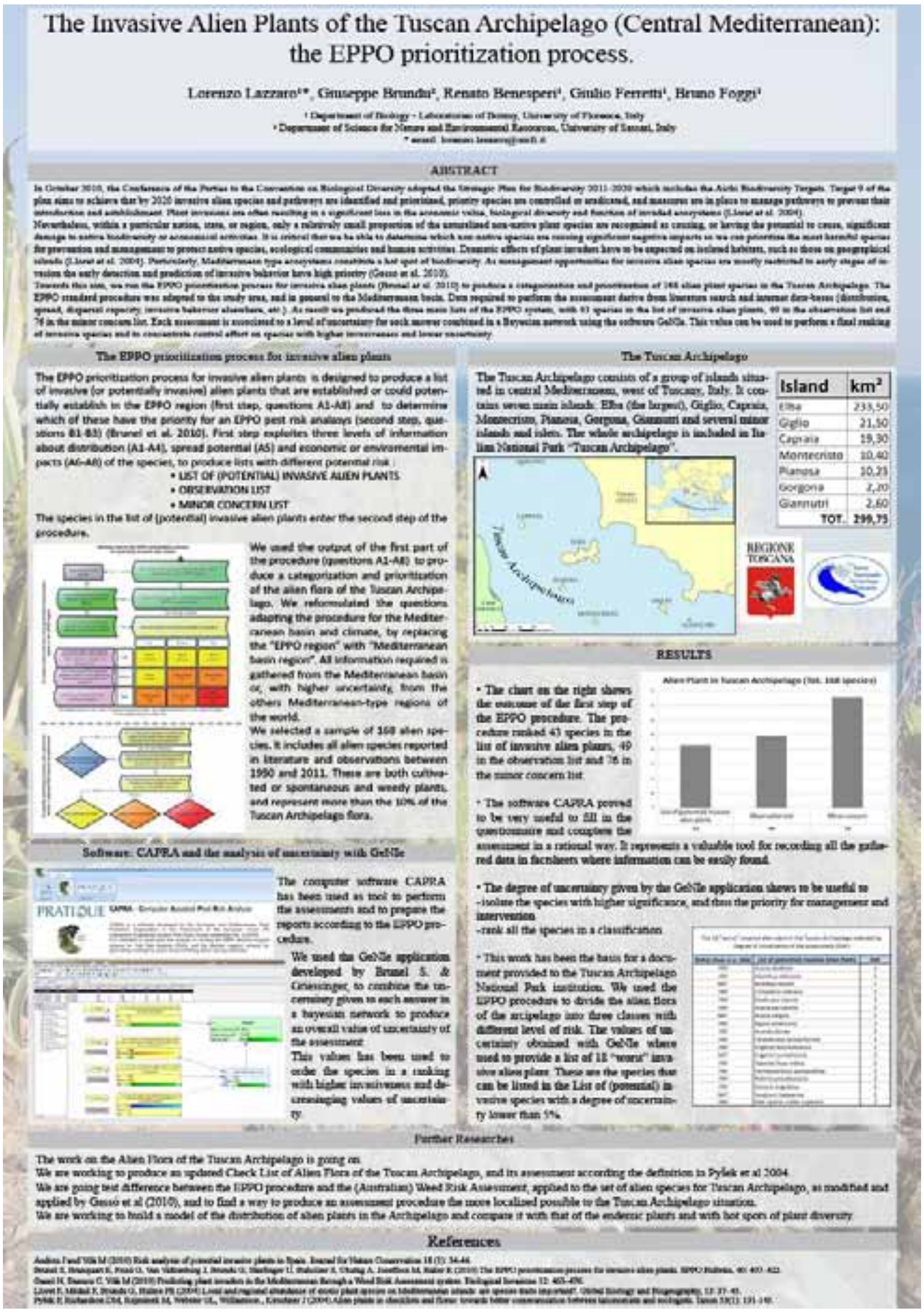
5.1 The Invasive Alien Plants of the Tuscan Archipelago (Central Mediterranean): the EPPO Prioritization Process



The Invasive Alien Plants of the Tuscan Archipelago (Central Mediterranean): the EPPO prioritization process.

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ABSTRACT

The EPPO prioritization process for invasive alien plants

The EPPO prioritization process for invasive alien plants is designed to produce a list of invasive (or potentially invasive) alien plants that are established or could potentially establish in the EPPO region (first step, questions A1-A8) and to determine which of these have the priority for an EPPO pest risk analysis (second step, questions B1-B3) (Brunel et al. 2010). First step exploites three levels of information about distribution (A1-A4), spread potential (A5) and economic or enviromental impacts (A6-A8) of the species, to produce lists with different potential risk :

- LIST OF (POTENTIAL) INVASIVE ALIEN PLANTS
- OBSERVATION LIST
- MINOR CONCERN LIST

The species in the list of (potential) invasive alien plants enter the second step of the procedure.

We used the output of the first part of the procedure (questions A1-A8) to produce a categorization and prioritization of the alien flora of the Tuscan Archipelago. We reformulated the questions adapting the procedure for the Mediterranean basin and climate, by replacing the "EPPO region" with "Mediterranean basin region". All information required is gathered from the Mediterranean basin or, with higher uncertainty, from the others Mediterranean-type regions of the world.

We selected a sample of 168 alien species. It includes all alien species reported in literature and observations between 1950 and 2011. These are both cultivated or spontaneous and weedy plants, and represent more than the 10% of the Tuscan Archipelago flora.

Software: CAPRA and the analysis of uncertainty with GeNIe

The computer software CAPRA has been used as tool to perform the assessments and to prepare the reports according to the EPPO procedure.

We used the GeNIe application developed by Brunel S. & Griessinger, to combine the uncertainty given to each answer in a bayesian network to produce an overall value of uncertainty of the assessment.

This values has been used to order the species in a ranking with higher invasiveness and decreasinging values of uncertainty.

The Tuscan Archipelago

The Tuscan Archipelago consists of a group of islands situated in central Mediterranean, west of Tuscany, Italy. It contains seven main islands: Elba (the largest), Giglio, Capraia, Montecristo, Pianosa, Gorgona, Giannutri and several minor islands and islets. The whole archipelago is included in Italian National Park "Tuscan Archipelago".

Island	km²
Elba	233,50
Giglio	21,50
Capraia	19,30
Montecristo	10,40
Pianosa	10,25
Gorgona	2,20
Giannutri	2,60
TOT.	299,75

RESULTS

- The chart on the right shows the outcome of the first step of the EPPO procedure. The procedure ranked 43 species in the list of invasive alien plants, 49 in the observation list and 76 in the minor concern list.

- The software CAPRA proved to be very useful to fill in the questionnaire and complete the assessment in a rational way. It represents a valuable tool for recording all the gathered data in factsheets where information can be easily found.

- The degree of uncertainty given by the GeNIe application shows to be useful to
-isolate the species with higher significance, and thus the priority for management and intervention
-rank all the species in a classification

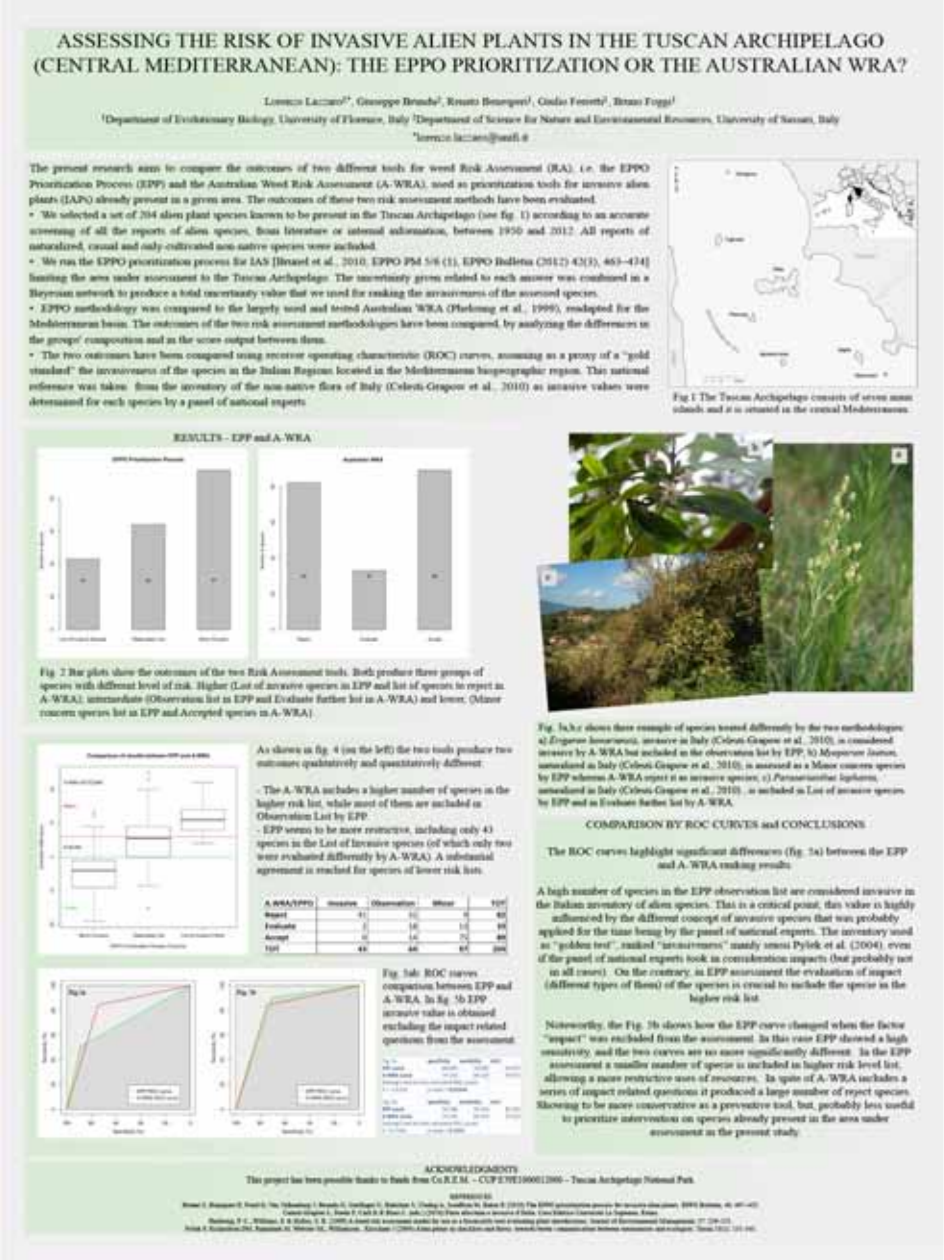
- This work has been the basis for a document provided to the Tuscan Archipelago National Park institution. We used the EPPO procedure to divide the alien flora of the arcipelago into three classes with different level of risk. The values of uncertainty obtained with GeNIe where used to provide a list of 18 "worst" invasive alien plant. These are the species that can be listed in the List of (potential) invasive species with a degree of uncertainty lower than 5%.

Further Researches

The work on the Alien Flora of the Tuscan Archipelago is going on.
We are working to produce an updated Check List of Alien Flora of the Tuscan Archipelago, and its assessment according the definition in Pyšek et al 2004.
We are going test difference between the EPPO procedure and the (Australian) Weed Risk Assessment, applied to the set of alien species for Tuscan Archipelago, as modified and applied by Gassó et al (2010), and to find a way to produce an assessment procedure the more localized possible to the Tuscan Archipelago situation.
We are working to build a model of the distribution of alien plants in the Archipelago and compare it with that of the endemic plants and with hot spots of plant diversity.

References

5.2 Assessing the risk of invasive alien plants in the Tuscan Archipelago (Central Mediterranean): the EPPO prioritization or the Australian WRA?



ASSESSING THE RISK OF INVASIVE ALIEN PLANTS IN THE TUSCAN ARCHIPELAGO (CENTRAL MEDITERRANEAN): THE EPPO PRIORITIZATION OR THE AUSTRALIAN WRA?

5.3 Priority Invasive alien plants in the Tuscan Archipelago (Italy): comparing the EPPO prioritization scheme with the Australian WRA

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In preparation for Biological Invasions

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Abstract

Biological invasions are a global phenomenon that threatens biodiversity and human economy. Many risk assessment frameworks have been developed, aiming to identify and/or prioritize potential future invasives, but an European standard risk assessment method does not yet exist. The EPPO priorization process recently developed could represent a unified and adaptable tool in the risk assessment of invasive alien species in Europe. With this work we aimed to test with test for the first time the EPPO priorization process method on 212 alien plant species thriving in a protected area (the Tuscan Archipelago National Park, Central Italy); moreover we aimed to compare the results of the EPP with those of the Australian Weed Risk Assessment, testing both methods with the categorization done at national and local level by panel of experts. Our results suggest that both the methods are good in the prediction of invasiveness of species already present in the Tuscan Archipelago. However the total numbers of species included in the highest risk categories by the two assessed methods differs greatly, with the EPPO priorization process including less species in the highest risk class. The two methods differ in the ability to detect correctly the invasive species, with the Australian Weed Risk Assessment more conservative but at cost of more species in the highest risk class. These results suggest that the choice of the risk assessment to use should be driven by the aim of the assessment, i.e. distinguishing between predictive and prioritization aims.

Keywords

Mediterranean islands, risk assessment, prioritisation of alien plants

Introduction

Biological invasions are a global phenomenon that threatens biodiversity and human economy, and few, if any, ecosystems are free from alien species (Catford et al. 2012), including protected areas (e.g., Foxcroft et al. 2013). The impacts of non-native species generally increase if the species establish themselves and spread in their new environment (i.e., if they become invasive sensu Blackburn et al. 2011), but non-native species can have impacts even when they are not established or widespread (Ricciardi and Cohen 2007; Ricciardi et al. 2013; Jeschke et al. 2014). As a consequence, a central focus of invasion biology has been to try to understand the invasion process and to distinguish between those few alien species that cause harm and those that do not, as a way to characterise the risk associated with alien species (Rejmanek et al. 2005). Both black-list and Green list approaches can be used in this concern (Dehnen-Schmutz 2011).

There is a systematic relationship between damages caused by alien species and a set of conditions knowable in advance. The former is the motivation for prioritising efforts, and the latter is the motivation for using risk assessment (RA) tools (Whitney and Gabler 2008; Leung et al. 2012).

In this framework in October 2010, the Conference of the Parties to the Convention on Biological Diversity adopted the Strategic Plan for Biodiversity 2011-2020 which includes the Aichi Biodiversity Targets. Target 9 of the plan aims to achieve that by 2020 invasive alien species and pathways are identified and prioritized, priority species are controlled or eradicated, and measures are in place to manage pathways to prevent their introduction and establishment (http://www.cbd.int/sp/targets/).

Many risk assessment frameworks have been developed for plants (see Essl et al. 2011; Kumschick and Richardson 2013 and references therein), but an European standard risk assessment method does not yet exist in spite of an urgent need of a unified EU strategy on invasive alien species and of list of species of Union concern (*ex* Reg. (EU) 1143/2014). The European and Mediterranean Plant Protection Organization (EPPO) "pest risk analysis” (EPPO PRA) represent an useful tool, with its assessments having direct and legally binding consequences for invasive plant management (notification and eradication requirements) in the International Plant Protection Convention (IPPC) context (Essl et al. 2011). However it represents a long procedure still far for being suitable to be used broadly for prioritizing or assess large number of species in a relative short amount of time. However in the last years a rapid screening prioritization process was proposed by the EPPO, the EPPO priorization process (EPP) (Brunel et al. 2010; EPPO 2012). This approach could represent a unified and adaptable tool in the risk assessment of invasive alien species in Europe. However, up to now, while other older risk assessment schemes, such as the Australian Weed Risk Assessmente (A-WRA), have been applied on large number of plants and in many countries, the EPP has been tested yet much less frequently.

Therefore, the present research aims to: (1) test for the first time the EPP method on a significant number of alien plant species (212) thriving in a protected area (the Tuscan Archipelago National Park, Central Italy); (2) compare the results of the EPP with those of the Australian WRA; (3) compare the results of both methods with the categorization done at national and local level by panel of experts.

Methods

Study area

The research focuses on the alien flora of the "Tuscan Archipelago National Park" (TANP). This Italian protected area is located in the central Mediterranean Sea and consists of seven main islands: Elba (the largest), Giglio, Capraia, Montecristo, Pianosa, Gorgona, Giannutri and several minor islands and islets, with a total surface of about 230 km^2. The flora of the TANP counts 1,300 species (Arrigoni et al. 2003) and it includes a 1.2% of narrow endemic species. The landscape of the TANP is typically Mediterranean, dominated by sclerophyllous-evergreen forest with all its different stages such as high and low macchia, garrigues and discontinuous ephemeral grasslands. The human influence on the natural environment has started around 6,000 years b.p. and has become massive since the Roman times (2,400 years b.p.). In the last century, like many other Mediterranean islands, the TANP islands have undergone a deep change of their landscape and land-uses, from agro-forestry to tourism activities. This change has been also a major driver of the introduction of alien plants, mainly as ornamental (Lazzaro et al. 2014).

Filtering out species present in the target region

A group of 212 alien plant species was selected from the alien flora of the TANP (Lazzaro et al. 2014). It includes 140 casual or naturalized species and 72 only-planted species. The selection includes the causal and naturalized species retrieved in the wild after the 1950 and species that can be found cited after 1950 as only-planted in the several contributions on the TANP, i.e. Montecristo (Paoli and Romagnoli 1976), Elba Island (Fossi Innamorati 1983, 1989, 1991, 1994, 1997), the contributions by Baldini (1998, 2000, 2001) for Giglio, Pianosa and Giannutri and by Foggi et al. (2001) for Capraia.

Defining the reference for the invasive status (estimate of invasivness)

The invasive status of all the selected 212 species, has been already assessed by panels of experts both at national (Celesti-Grapow et al. 2009; 2010a; 2010b). [Riferimento alla tabella nei materiali supplementari] and at TANP level (Lazzaro et al. 2014). Both the assessments were adopted as an estimate of invasiveness of the species aiming to benchmark the performances of the other two methods assessed (EPP and A-WRA, see below).

For both the estimates we built a quantitative and a binary score. Quantitative score in Italy (Rank Italy) was calculated adopting the following formula:

$$\text{Rank Italy} = \frac{3 * \Sigma INV \; + 2 * \Sigma NAT + 1 * \Sigma CAS}{63}$$

where ΣINV, ΣNAT, ΣCAS are the number of regions in which the species has been assessed as invasive, naturalized and casual respectively. The result of the sum is then divided by 63, which would be the maximum score for a species given as invasive in all the 21 Italian regions, to standardize the score between 0 and 1. For the ROC analyses this score was transformed in a binary score (Binary Italy) considering as invasive those species that had been assessed as invasive in at least one Italian region.

A similar method was adopted to create a quantitative score of invasiveness for the TANP (Rank TANP), calculated adopting the following formula:

$$\text{Rank TANP} = \frac{3 * \Sigma INV_{TANP} \; + 2 * \Sigma NAT_{TANP} + 1 * \Sigma CAS_{TANP}}{24}$$

where ΣINV_{TANP}, ΣNAT_{TANP}, ΣCAS_{TANP} are the number of islands of the Archipelago where the species has been assessed as invasive, naturalized and casual respectively. The result of the sum is then divided by 24 which would be the maximum score for a species given as invasive in all the 7 islands + minor islets, to standardize the value between 0 and 1. Again for the ROC curves analyses the value was transformed in a binary output (Binary TANP) considering as invasive the species that had been assessed as invasive in at least one island.

Nevertheless, it is important to notice that the check list of TANP give the invasive status of the species mostly according to Pyšek et al. (2004)

definition, i.e. it doesn't take into full account the potential of actual negative impacts.

Gathering information on the alien species

For each species the data required to carried out the two risk assessments was gathered from several sources, including scientific literature, local reports on impact or distribution, personal observations in the Tuscan Archipelago and information from internet databases like USDA Plants database (http:// plants.usda.gov), Global Compendium of Weeds (http://www.hear.org/gcw), Global Invasive Species Database (http://www.issg.org/database/welcome), Weeds in Australia (http:// www.weeds.gov.au), DAISIE site: http://www.europe-aliens.org/default.do, Cal-IPC database (http://www.cal-ipc.org/ip/inventory/weedlist.php); Australian Flora Online (http://www.environment.gov.au/biodiversity/abrs/online-resources/flora/main/index.html) or others Risk Analysis available such as Tasmanian Weed Risk Assessment page from Department of Primary Industries, Parks, Water and Environment: http://www.dpiw.tas.gov.au/inter.nsf/WebPages/SWEN-7S74GE?open. and the EPPO database PQR (EPPO 2014; http://www.eppo.int/DATABASES/pqr/pqr.htm).

Running the EPPO priorization process for invasive alien plants and the Australian WRA

The EPPO priorization process (EPP) is based up on relatively simple but robust criteria, accounting for biogeography, distribution, invasiveness elsewhere, (potential) impacts (both in natural or semi-natural habitats and agricultural systems), and management efficiency (Brunel et al. 2010; EPPO 2012). The broad idea behind it is to select those species for which a Pest Risk Analysis (PRA) constitutes an adequate tool. It was recently revised by the PRATIQUE EU project becoming more efficient, user-friendly (Baker et al. 2009; Steffen et al. 2012) and supported by the Computer Assisted Pest Risk Analysis software (i.e., CAPRA). The EPP is designed (i) to produce a list of invasive alien plants that are established or could potentially establish in the area under assessment and (ii) to determine which of these have the highest priority for an EPPO Pest Risk Analysis (PRA). As such, it could be used also to rank invasive species in three main categories (minor concern, observation list, invasive). Furthermore, the software for Bayesian networks GeNie which is embedded in CAPRA, giving the possibility to record and track uncertainty in the assessment, gives further possibility of ranking within the 3 main produced groups (minor concern, observation list, invasive plants). Indeed GeNIe produces a distribution of probability of the inclusion of the species in each group (with total probability being equal to 1). It should be kept in mind that the process is designed to perform rapid assessments, and to provide structured and traceable information on alien species, and cannot be considered a substitute to a full PRA. As our aim was to prioritize the alien flora of the Tuscan Archipelago, we run only the first part of the procedure (questions A1-A8). We reformulated the questions adapting the procedure for the Mediterranean basin and climate, by replacing the “EPPO region” with “Mediterranean basin region". Giving the uncertainty value to the questions and to the whole assessment we followed EPPO (2012) and the above indications. We considered a low level of uncertainty for those species whose ecology was well known and studied at the level of the Tuscan Archipelago. On the contrary, we considered a medium to high-value in the other cases (i.e. information available at the scale of the Mediterranean basin, or only for other Mediterranean-type regions of the world). The values of probability provided by GeNie were used to assign a score to each species (EPP score) aiming to rank the species. The EPP score of each species is given by the formula:

$$EPP\ score\ = 3 * INV + 2 * OBS + 1 * MC$$

where INV; OBS and MC represent the probability to be in the list of Invasive Species; Observation list or Minor Concern list according to GeNie results.

The Australian Weed Risk Assessment (A-WRA) developed by Pheloung (1999) has been widely used in Australia and outside, including Europe (see Hawaii and Pacific islands: Daehler and Carino 2000; Daehler et al. 2004; Italy: Crosti et al. 2007; Bonin Islands: Kato et al. 2006; Czeck Republic: Křivánek and Pyšek 2006; Mediterranean basin Gassó et al. 2009; Spain: Andreu and Vilà, 2010). According to Gordon et al. (2008a, 2008b, 2010) the WRA accuracy in different geographic regions has found to be generally high. The A-WRA is designed as a predictive tool with the aim to assess the risk of invasion related to the introduction of alien species. It allows to assess if the new species should be rejected or accepted for import, or if further evaluation is needed in case of intermediate or unknown risk. This is done answering to a set of 49 questions regarding biogeography, undesirable plant attributes and biology/ecology (Pheloung et al. 1999) allowing to produce a scoring of the species (A-WRA score). A score higher than 6 identifies a species likely to be of high risk and therefore to be rejected for import; a score equal or lower than zero implies to accept the species for import and intermediate scores (between 1 and 6) mark species that require further evaluations. Following Gassó et al. (2009) we modified some of the questions to adapt the scheme for Mediterranean basin. Question 2.01 was changed from ‘‘Species suited to Australian climates’’ to ‘‘Species suited to Mediterranean climates’’. And question 5.03: ‘‘Nitrogen fixing woody plant’’ to ‘‘Nitrogen fixing plant’’, to include the non- woody nitrogen fixing plants, which represent an important group in Mediterranean flora.

In conclusion, each of the selected 212 species was evaluated using both the European and Mediterranean Plant Protection Organization Prioritization standard method [PM 5/6(1)] (EPPO 2012) and the A-WRA, modified for the Tuscany archipelago.

Statistical analysis

The grouping outputs of the two assessments were compared by means of a contingency table, and the differences in proportion of species assessed in each risk group were investigated making use of a Chi-square test.

We studied the relationship between the EPP score and A-WRA score and the estimates for invasiveness in Italy and in TANP by means of linear regression models. The estimates were considered as response variables and the EPP and A-WRA scores were considered as the predictors. For both the estimates an arcsine transformation ($X=\text{arcsine}(\sqrt{(X)})$) was adopted aiming to normalize the residuals. Significance of the regression line was evaluated by means of analysis of variance studying sum of squares and degree of freedom against a F distribution.

The ability of the two methods to detect invasive species was tested again the binary estimates of invasiveness in Italy and in the TANP by means of receiver operating characteristic curves (ROC) (DeLong et al. 1988). This kind of analysis has been already tested as an useful tool for evaluating the performance of invasive species screening tests (e.g., Hughes and Madden 2003; Caley and Kuhnert 2006; Gordon et al. 2008a; Gassó et al. 2009).

The ROC curve technique allows to investigate the performance of a predictor against an observed response studying the proportion of true positives (i.e., *sensitivity*) against the proportion of true negatives (i.e., *specificity*) across a range of cutoff points (Gassó et al. 2009). The sensitivity of a RA method refers to the ability of the method to correctly identify those plant species that are invasive. On the contrary, the specificity of a RA method refers to the ability of the method to correctly identify those plant species that are not invasive. Following De Long et al. (1988) the area under the curve (AUC) is considered as a recommended index of accuracy of the model, representing the capability of the model to give the correct order for the scores of positive and negative cases (i.e. for both the A-WRA and the EPP a higher value for invasive species than not-invasive ones). The AUC of the curve vary from 0.5 (random choice of the model) to 1 (perfect matching with the response). Thus the closer the area under the ROC curve is to one, the better the screening tool's ability to differentiate between the two groups (Lasko et al. 2005), catching *all* (high sensitivity) and *only* (high specificity) the invasive species and discarding non invasive ones.

We study with ROC curves both the capability to detect invasive species as a binary output (EPP binary and A-WRA binary) and the scoring outputs (EPP score and A-WRA score) of the two methods. The discrete outcome of the two outcomes was expressed as binary variable 1/0 (i.e. 1 if the species was assessed in the High risk level and 0 if the species was assessed in a lower group according to table1). In both the cases while a binary "gold test" is necessary to use ROC curves we use a comparison the binary assessment of invasiveness in Italy and TANP (Binary Italy and Binary TANP).

To compare the curves and the differences in the area under the curve (AUC) we used the DeLong's test for two correlated ROC curves (DeLong et al. 1988).

Finally, we used GLMs to test the presence of possible correlation between the EPP (binary) and A-WRA (binary) and the year of first record in the TANP, the life span and growth form of the assessed alien plants. Furthermore, one more GLM, was fitted to test whether the agreement of the two methods, in including or not the species in the group of higher risk level was influenced by year of first record, life span and growth form. Binomial distribution of the error was adopted in the GLMs and significance of the terms was evaluated by means of analysis of deviance with degree of freedom against Chi-square distribution.

The ROC curves and the test between AUC's were performed using the *pROC* package ver. 1.6.0.1 (Robin et al. 2011) of the R software ver. 3.1.2 (R Core Team, 2014). The Chi-square tests, linear models and GLMs were performed using the R software ver. 3.1.2.

Results

Both RA procedures (EPP and A-WRA) provide a three-list categorization of the assessed alien plants, with an increasing level of risk (tab 1 and figure 1), but with significant differences (see Tab 1; Chi-squared = 22.4986, DF = 2, p-value <0.0001).

Level of Risk	EPP lists	A-WRA lists
HIGH	List of Invasive Species (47)	Reject Species List (88)
MEDIUM	Observation List (66)	Evaluate Further List (35)
LOW	Minor Concern List (99)	Accept Species List (89)

Table 1 Distribution of the 212 alien plant of the Tuscan Archipelago in the three classes of invasion risk according to the EPP and the A-WRA methods.

The EPP scheme ranked 47 species in the List of invasive species, 66 in the Observation List and 99 in the Minor Concern List. On the contrary, according to the A-WRA 88 species have to be rejected, 35 to be evaluated further and 89 can be accepted.

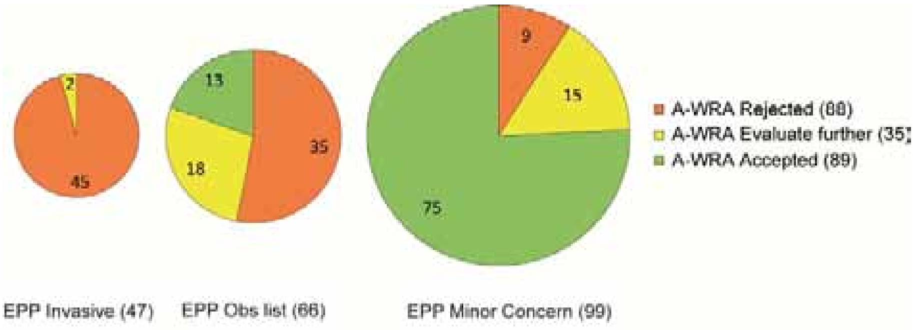

Fig. 1 Distribution of the 212 species in the three classes of EPP ("cakes") and A-WRA (colors). Radius of the circles is proportional to number of species according EPP assessment and surface of the slices represent proportion of species according A-WRA assessment.

Noteworthy, the total numbers of species included in the highest risk categories by the two assessed methods differs greatly (47 vs 88). The EPP included less species in the List of Invasive species and more species in the Observation List when compared to the A-WRA.

Figure 1 shows a qualitative comparison of the way the two assessments treat the species. Main differences were in the assessment of the species in the higher and intermediate risk lists. About high risk lists, the two outcomes showed a good agreement for a group of 45 species assessed in the highest risk group by both the procedures. However they showed a high disagreement for the other cases. In fact 35 and 9 species included in the invasive species by A-WRA were included respectively in the Observation List and Minor Concern List by EPP. Only two species (i.e the hybrid *Vitis riparia* x *Vitis rupestris* and *Paraserianthes lophantha*) assessed as invasive by EPP were assessed as intermediate risk by the A-WRA. The intermediate risk lists were again quite different. Only 18 species were assessed in the intermediate risk lists by both the procedures, whereas 13 species included in the observation list by the EPP were included in the lowest list by the A-WRA. Finally 75 species were included by both the procedures in the lowest level group.

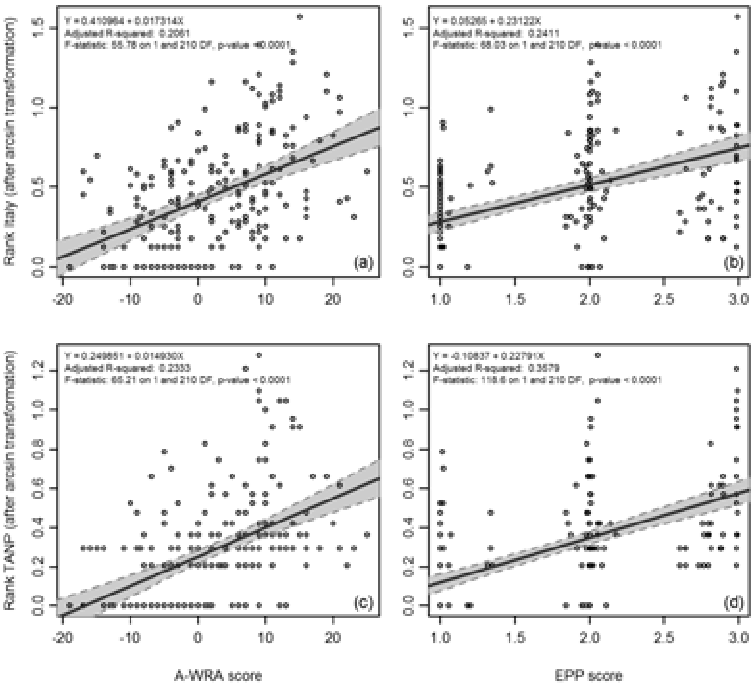

Fig. 2 Correlation plots for estimate of invasiveness in Italy (Rank Italy) with A-WRA score (a) and EPP score (b); and for estimate of invasiveness in TANP (Rank TANP) with A-WRA score (c) and EPP score (d). Red lines represent the fitted regression lines, whose statistics are indicate in the plots. Grey shadow polygons represent 95% confidence interval of regression line. An arcsine transformation was applied to the variables Rank Italy and Rank TANP

A-WRA score ranged from -19 (*Butia capitata*) to 25 (*Datura wrigthii*). EPP score ranged from 1.00 (many species – see Table xxx in supplementary matherials) to 2.99 (*Ailanthus altissima*). The linear regression models showed that both EPP and A-WRA scores correlate positively and significantly with the quantitative estimates of invasiveness for Italy and the TANP (all regression's P<0.0001; Fig. 2). Adjusted R-squared were generally good, ranging from 0.20 to 0.35. Particularly EPP showed a greater Adjusted R-squared than A-WRA in both the comparisons.

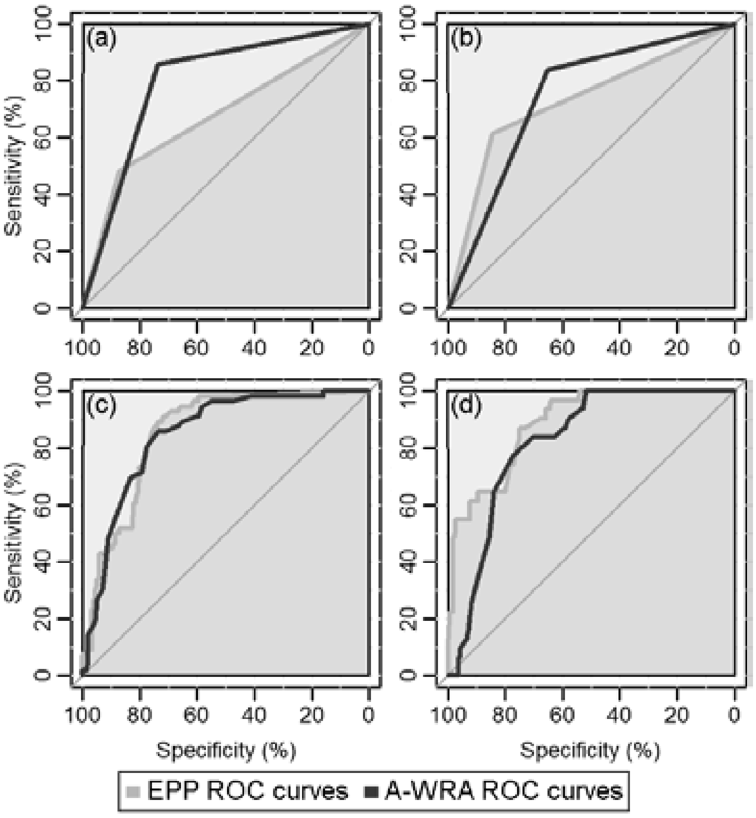

Fig. 3 ROC curves describing the performance of the EPP (light curve) and A-WRA (dark curve) to detect invasive species. In (a) and (b) the binary output of EPP and A-WRA is compared with the binary estimate of invasiveness respectively in Italy and TANP; in (b) and (c) the quantitative scoring output of EPP and A-WRA is compared again with the binary estimate of invasiveness respectively in Italy and TANP. Sensitivity represents the proportion of true positives (Number of true positive assessment)/(Number of all positive assessment). Specificity represents the proportion of true negatives (Number of true negative assessment)/(Number of all negative assessment).

Comparing the two binary outputs with the Binary Italy AUC were significantly different with A-WRA showing better performances (AUC for EPP: 67.70, and A-WRA: 79.72, Z = -3.1715, p-value = 0.001517). It is noteworthy that the two methods showed opposite behavior in term of sensitivity and specificity with EPP showing a high specificity and a low sensibility, whereas the contrary happen for the A-WRA, as observed above (Fig 3a; table 2). The AUCs in case of binary outcomes compared to Binary TANP were not significantly different according to the DeLong's test (AUC for EPP: 72.91, and A-WRA: 74.53, Z = -0.3437, p-value = 0.7311). However again a quite different weight of sensitivity and specificity between the two models could be detected (Fig 3b; table 2). EPP binary showed a high specificity and a low sensibility, whereas the contrary happen for the A-WRA.). This indicating that generally EPP showed a good capability to discard not invasive species, but on the other hand it discarded some invasive ones. Whereas A-WRA tends to include some non-invasive species in the reject species list, but had a good capability to not discard invasive species.

	AUC (%)	Cutoff	Specificity (%)	Sensitivity (%)	Observed response (gold test)	DeLong's test
EPP binary	67.70	0.50	87.17	48.21	Italy binary	Z = -3.1715, p-value = 0.0015
A-WRA binary	79.72	0.50	73.72	85.71		
EPP binary	72.91	0.50	84.53	61.29	TANP binary	Z = -0.3437; p-value = 0.7311
A-WRA binary	74.53	0.50	65.19	83.87		
EPP score	85.51	1.99	73.72	89.29	Italy binary	Z = 0.2782, p-value = 0.7809
A-WRA score	84.77	5.50	73.72	85.71		
EPP score	88.77	2.00	75.14	87.10	TANP binary	Z = 2.3438, p-value = 0.0190
A-WRA score	82.21	8.50	77.35	77.42		

Table 2 Results of ROC curves analysis of the EPP and the A-WRA outcomes, compared with the invasiveness estimated for Italy and the Tuscan Archipelago. AUC= Area Under the Curve. DeLong’s test p values are reported for comparable pairs of ROC curves.

Different results can be obtained when comparing the AUC and ROC curves on EPP and A-WRA scores. When comparing the scores with Binary Italy AUC appears not significantly different (AUC for EPP: 85.51, and A-WRA: 84.77, Z = 0.2782, p-value = 0.7809), and the shapes of the curves in terms of specificity and sensitivity appear comparable (Fig 3c; table 2). AUC are of curves obtained with the comparison with Binary TANP appeared significantly different (AUC for EPP: 88.77, and A-WRA: 82.21, Z = 2.3438, p-value = 0.01909), with a higher performances of EPP scores in term of sensitivity (Fig 3d; table 2).

	Term	Df	Deviance	Residual Df	Residual Deviance	P values
EPP binary	Growth form	7	14.94	204	209.38	0.03676*
	Life Span	2	4.20	202	205.17	0.12211
	Year of first record	1	1.11	201	204.06	0.29138
	Residuals	211	224.32			
A-WRA binary	Growth form	7	18.10	204	270.31	0.01151*
	Year of first record	1	6.00	203	264.31	0.01426*
	Life Span	2	3.94	201	260.37	0.13929
	Residuals	211	288.42			

Table 3 Analysis of deviance table for GLMs on invasive output of EPP and A-WRA assessments. Year of first record was centered at 1808, which is the smallest value. DF = Degree of Freedom. * marks significant terms at P<0.05.

Term	Df	Deviance	Residual Df	Residual Deviance	P values
Life Span	2	9.4917	209	212.29	0.008688**
Year of first record	1	4.3767	208	207.91	0.036433*
Growth form	7	8.8741	201	199.03	0.261822
Residuals	211	221.78			

Tab 4 Analysis of deviance table for GLM on agreement of EPP and A-WRA assessments on the invasive output. Year of first record was centered at 1808, which is the smallest value. DF = degree of Freedom. * marks significant terms at P<0.05; ** marks significant terms at P<0.01

According to GLMs on the discrete outcome both assessment suffer from some degree of dependence from one or more of the tested explanatory variables. EPP assessment appeared significantly correlated with growth form, while A-WRA appeared correlated both with growth form and year of first record of the species (Tab 4). Analyzing the proportion of reject outputs for the different growth forms it appears that for EPPO the main deviation from the mean proportion occurs for geophyte, scrubs and forbs (fig 4a), for WRA it occurs for the same growth forms but also for trees (fig 4b). Moreover WRA assessments seems to suffer from a dependences form year of first record with a greater proportion of invasive species from those reported around middle 1900 (Fig 5).

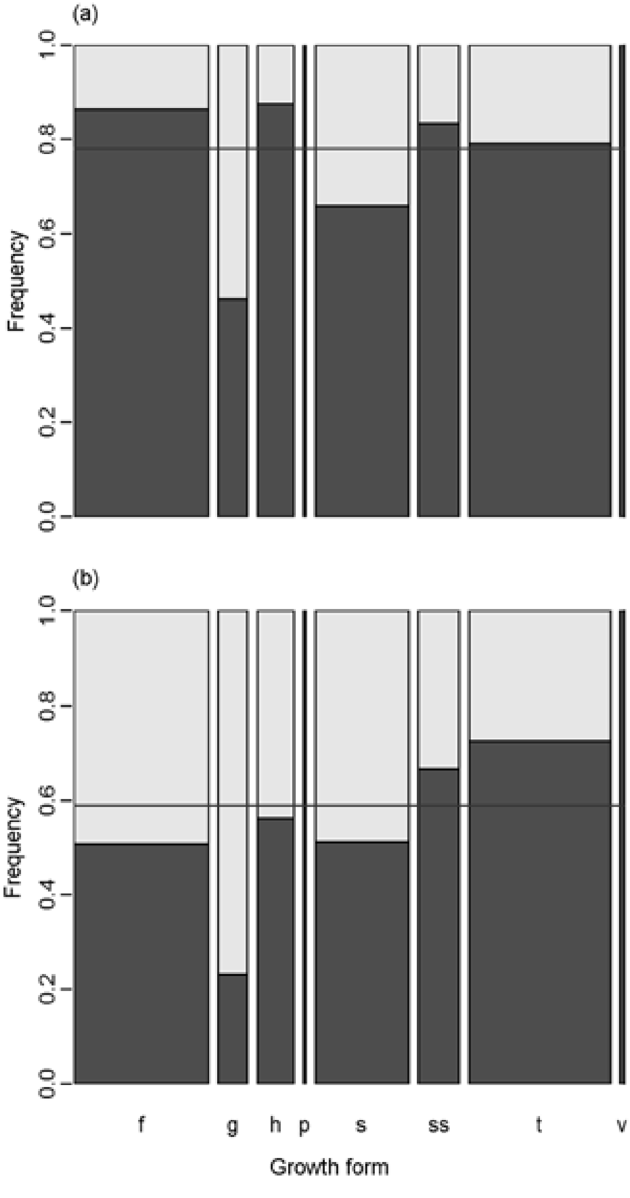

Fig 4. Spine bar plot showing the conditional relative frequencies of species assessed in the high-risk group by EPP (a) and A-WRA (b) according to their growth form. Dark shadow part of the bar correspond to the proportion of species assessed as not invasive and light shadow correspond to proportion of species assessed as invasive. Black horizontal lines represent the mean proportion of species assessed by the method in the high-risk group. Widths of the bars corresponds to the relative frequencies of species according to life forms f=forbs, g=grass, h=herbal, p=parasitic, s=shrub, ss=subshrub, t=tree, v=vines.

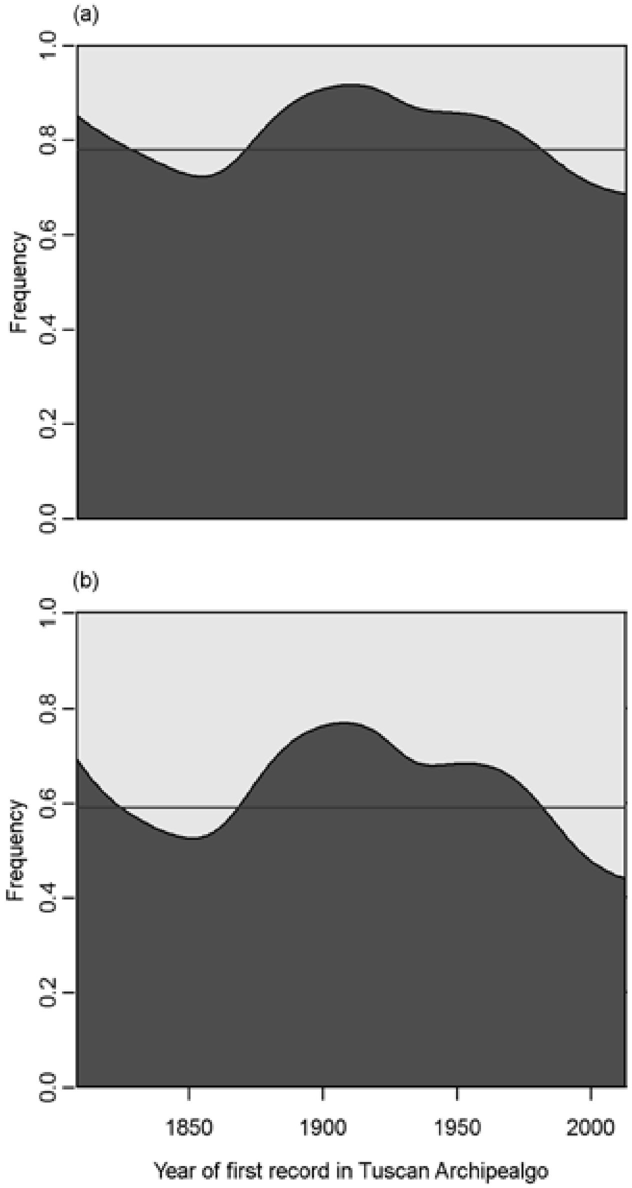

Fig 5. Conditional densities plot showing the conditional distribution species assessed in the high-risk group by EPP (a) and A-WRA (b) according to year of first record. Dark shadow part of the graph corresponds to the proportion of species assessed as not invasive and light shadow correspond to proportion of species assessed as invasive. According to GLM results only in (b) proportion of species assed as invasive is significantly affected by the year of first record in Tuscan Archipelago.

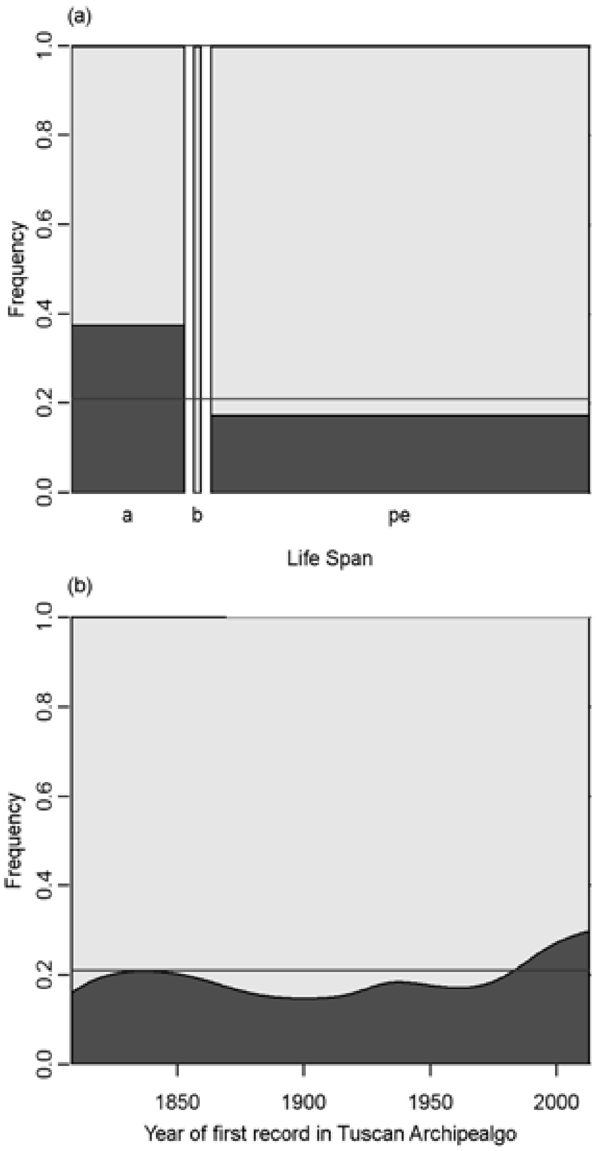

Fig 6 Spine bar and conditional density plots showing the conditional relative frequencies of agreement of EPP and A-WRA in including the species in the highest risk group according to their life span (a) and year of first introduction in the Tuscan Archipelago (b). Dark shadow part of the plot correspond to the proportion of species for which the two assessment did not agree and the light shadow correspond to proportion of species for which they did. Horizontal lines represent the mean proportion of species for which the two assessment agreed. Widths of the bars in 6a corresponds to the relative frequencies of species.

The agreement across the two methods appeared influenced by the growth form and by life span of the species (Tab 5). The main proportion of "disagreement" appear related to annual species, and to those species recorded after the 2000 (Fig 6). It is noteworthy that most of the disagreement is given by species assessed in the high-risk group by A-WRA but not by EPP (43 species versus only 2 in the opposite case).

Discussion

The aim of the present work was to identify priority invasive alien species within a set of alien plant species of the Tuscan Archipelago and to benchmark two different methods, i.e. the EPPO Prioritization method and the Australian Weed Risk Assessment. The ranking produced by these two methods resulted both positively correlated with the existing estimates of invasiveness for the evaluated species both at national and local level (Italy and TANP). However, these two methods showed some differences in the ranking and classification of the alien species. Particularly, rather than differences across AUC of the ROC curves, we detected important differences in term of sensibility and sensitivity.

The data-mining necessary to produce an outcome requires basically the same kind of information. For both methods information on introduction and invasive history of the species under assessment, general information on the spread potential and on the (potential) impacts has to be retrieved. However the more specific approach of the A-WRA requires a very detailed search of specific plant traits for some of the requested information, whereas for the EPP a more generalist approach is required, allowing the assessor to save some time. On the other hand aiming to answer a specific question is more difficult to lose some important literature, therefore an intense effort have to be used also for the EPP production.

The importance of climate matching have been already stressed as an important factor in the establishment and success of alien species (Kumschick and Richardson 2013). One of the issues raised for A-WRA is the lack of a reference for the use of an appropriate method to assess climate matching, while in EPP several maps are provided as a reference for the choice of climate matching (EPP 2012). In our case, as already done in other cases (see Gordon et al. 2008a; 2008b; 2010) we followed the Australian recommendation to adopt highest scores for the climate matching questions where no computer analysis of climate matching is carried out.

EPP is a quite recent risk assessment method (Brunel et al. 2010; EPPO 2012) and up to now it was not possible to find any work testing the method on a wide set of alien species, whereas more literature is available on the A-WRA (Gordon et al. 2008a; 2008b; 2010; McGregor et al. 2012; Speek et al. 2013).

The correlation of the A-WRA ranking with the existing estimates of invasiveness for Italy and the TANP is generally higher than the results obtained by Speek et al. (2013). It is noteworthy that explained variance was higher for EPP ranking. Speek et al. (2013) assume that the low correlation of A-WRA with the estimates of invasiveness was mainly due to the low capability of A-WRA to catch the impacts of the species. This could also explain the differences in our results. In fact in A-WRA impacts are strictly categorized by specific questions, while it has been discussed that impacts of invasive species can greatly vary in different region and can be differently interpreted (Speek et al. 2013). In case of EPP the definition of possible impacts is more general, thus likely to be more appropriate to catch presence of impacts across different region and contexts.

AUC of the ROC curves were generally lower for the binary outcome of the two methods than when studying the ranking outputs. This highlight that the threshold for including or not a species in the invasive species list should be carefully evaluated before applying the method. Indeed the AUC values of ROC analyses on quantitative scoring output of A-WRA are consistent with those reported in other case of studies (Gordon et al. 2008a), where was found to range from 0.82 to 0.99. Gordon et al. (2008a) underlined the importance for stakeholders and land managers to set an appropriate cutoff value according to the aim of the assessment. Setting an appropriate cutoff value is in fact affecting not only the general efficiency of the assessment, but more in deep the relationship between specificity and sensitivity of the method.

Following the default cutoff for A-WRA and the default priorization decision structure on EPP leaded to very different results in the identification of invasive species. This is underlined by the differences in ROC curves studying the binary output of the assessments. EPP has generally a higher specificity, whereas A-WRA has higher sensitivity, also in case when the two AUC are comparable. These differences could be related to the different aim of the two tools, with the EPP being mainly a priorization tool and the A-WRA a predictive tool (Essl et al. 2011). Considering the aim of prioritize the available resources on a set of alien species it would be preferable to obtain a restricted set of specie with high risk level, so as not to waste resources in a too wide set of species. In this case the EPP is more conservative in the direction of avoid "false positive", leading to a small set of high risk species, whereas in the A-WRA it has been shown a tendency to commit relatively more "false positive" than "false negative" (Koop et al. 2011). On the other hand the higher performances in sensitivity by A-WRA appear consistent with the aim to avoid new introductions of potentially harmful species. In this perspective, it appear reasonable to pay the cost of some false positives if we are able to maximize the catching of true positives (Daehler et al. 2004). This consideration are consistent to the results obtained by McGregor et al. (2012), where an extremely high accuracy in detecting g invasive species was reflected in a high number of false positives suggesting that accuracy comes at a cost of rejecting potentially useful species.

In both the assessments, the score is potentially suitable to produce a classification of the species in order of higher risk of invasion, but the two score are underlining different meanings. The A-WRA score is readable as a proxy of the potential risk related to the species. As already said this approach con be affected by the choice of what is an undesirable traits or which are the impacts produced by the species. On the other hand, the EPP score is related to the assessment uncertainty. Thus according to EPP plants are classified according to possible impacts but the final value is influenced by the available information on the species. Beside that, the definition of impacts is more general than in A-WRA, the assessment could also be updated in case new relevant information is available for a certain species, thus leading to a more flexible approach.

The presence of bias in the screening methods have been largely investigated in past. Both the methods appear affected by the growth form of the species. It is not clear if this trend is more related to an effective differential risk according to the growth form or from a bias in the capability of the evaluation of impacts related to certain growth forms. The need for an unbiased evaluation of impacts is actually one of the main issues nowadays in the framework of IAS and risk assessments (Hulme et al. 2013; Jesche et al. 2014). Particularly according Hulme et al. (2013) life forms, strictly related to growth forms we studied, are one of the main source of bias in the evaluation IAS of impacts. For less dominant life forms, such as geophytes, impacts are probably less studied and more difficult to find. Moreover in case of A-WRA the possible outcome has been shown to be influenced by the time of record of the species (REF).

In conclusion, we found that both the methods are good in the prediction of invasiveness of species already present in TANP. However it looks that the more generalist approach of EPP was more efficient in our case. Particularly we confirmed that the choice of a certain cutoff is important giving the aim of the assessment procedure since the efficiency of the risk assessments can change quantitatively and qualitatively according to this choice. It appear clear that also the choice of the risk assessment to use should be driven by the aim of the assessment, i.e. distinguishing between predictive and prioritization aims. This work has been the basis for a document provided to the Tuscan Archipelago National Park institution aiming to prioritize the species already present on the area of study. Accordingly, we used the EPPO procedure to classify the alien flora of the Archipelago into three classes with different level of risk and the values of uncertainty obtained with GeNIe were used to provide a short list of 15 “worst” invasive alien plant.

Least but not last it is noteworthy that risk assessment procedures are strongly dependent on the available information on impacts and, as often pointed out, more information on non dominant species and multi-scalar approach in the study of impacts of IAS are urgently needed to be incorporate in the risk assessments.

Acknowledgments

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Appendix 1 Species used to test the EPP and the A-WRA. Life Span: a = Annual, b = biennal, pe = perennial. Life form: g = grass, f = forb, h = herbaceous, ss = subshrub, = shrub, t =tree, v= vines, p = parasitic.

Species	First Record In TANP	Life Span	Life form	EPP list	EPP score	WRA list	WRA Score
Abies nordmanniana (Steven) Spach	1964	pe	t	MC	1.000	Accept	-6
Abutilon theophrasti Medik.	2011	a	f	OBS	2.023	Reject	9
Acacia dealbata Link	1998	pe	t	INV	2.989	Reject	16
Acacia melanoxylon R. Br.	2011	pe	t	INV	2.789	Reject	15
Acacia provincialis A. Camus	1998	pe	t	INV	2.602	Reject	9
Acacia pycnantha Benth.	1973	pe	t	INV	2.891	Reject	13
Acacia saligna (Labill.) H.L. Wendl.	2012	pe	t	INV	2.760	Reject	7
Aeonium arboreum (L.) Webb & Berthel.	1974	pe	ss	OBS	1.907	Evaluate	2
Aeonium haworthii Salm-Dyck ex Webb & Berth.	2011	pe	ss	MC	1.000	Evaluate	4
Aesculus hippocastanum L.	1898	pe	t	MC	1.000	Accept	-7
Agave americana L.	1844	pe	ss	INV	2.980	Reject	7
Ailanthus altissima (Mill.) Swingle	1898	pe	t	INV	2.990	Reject	15
Albizia julibrissin Durazz.	1974	pe	ss	OBS	2.108	Evaluate	1.5
Allium cepa L.	1900	pe	h	MC	1.000	Accept	-1
Allium sativum L.	1975	pe	t	MC	1.000	Accept	-1
Alnus cordata (Loisel.) Loisel.	1964	pe	t	OBS	1.843	Accept	-5

Species	First Record In TANP	Life Span	Life form	EPP list	EPP score	WRA list	WRA Score
Aloe arborescens Mill.	1900	pe	h	MC	1.056	Accept	-7
Amaranthus albus L.	1847	a	f	OBS	2.010	Reject	10
Amaranthus blitoides S. Watson	2009	a	f	OBS	2.010	Reject	13
Amaranthus caudatus L.	2012	a	f	OBS	2.010	Evaluate	4
Amaranthus deflexus L.	1839	pe	f	OBS	2.010	Reject	11
Amaranthus graecizans L.	1870	a	f	OBS	2.000	Reject	11
Amaranthus hybridus L.	1900	a	f	OBS	2.000	Reject	16
Amaranthus retroflexus L.	1839	a	f	OBS	2.010	Reject	14
Anredera cordifolia (Ten.) Steenis	1974	pe	v	OBS	1.995	Evaluate	3
Antirrhinum majus L. subsp. majus	1901	pe	f	OBS	1.019	Accept	-3
Artemisia verlotiorum Lamotte	1839	pe	g	INV	2.881	Reject	12
Arundo donax L.	1981	pe	f	INV	2.980	Reject	12
Asclepias fruticosa L.	2010	pe	h	OBS	2.000	Evaluate	0
Asparagus falcatus L.	1997	pe	ss	MC	1.000	Evaluate	3.5
Austrocylindropuntia subulata (Muehlenpf.) Backeb.	2002	pe	ss	OBS	2.000	Evaluate	5
Avena byzantina K. Koch	2011	a	g	MC	1.000	Accept	-8
Avena sativa L. s.l.	1900	a	g	MC	1.000	Accept	-9
Beta vulgaris L. subsp. vulgaris	1870	b	f	MC	1.000	Accept	-4
Bidens pilosus L.	2011	a	f	INV	2.813	Reject	21
Bougainvillea spectabilis Willd.	1974	pe	v	MC	1.000	Accept	-14
Brassica napus L. subsp. napus	1867	a	f	OBS	2.000	Evaluate	0
Brassica oleracea L.	1839	pe	f	OBS	1.915	Accept	-15
Brassica rapa L. s.l.	1898	pe	t	MC	1.000	Evaluate	5
Broussonetia papyrifera (L.) Vent.	1844	a	f	OBS	2.178	Reject	6
Buddleja davidii Franch.	1974	pe	t	INV	2.602	Reject	11
Butia capitata (Mart.) Becc.	1980	pe	s	MC	1.000	Accept	-19
Callitropsis arizonica (Greene) D.P. Little	1882	pe	ss	MC	1.000	Accept	-14
Callitropsis glabra (Sudw.) D.P. Little	1975	pe	t	MC	1.000	Accept	-14
Callitropsis lusitanica (Mill.) D.P. Little	1964	pe	t	MC	1.000	Accept	-14
Callitropsis macrocarpa (Hartw.) D.P. Little	2010	pe	ss	MC	1.000	Accept	-14
Campsis radicans (L.) Bureau	1999	pe	t	OBS	1.873	Evaluate	2
Canna indica L.	2012	pe	t	OBS	1.946	Evaluate	1
Carpobrotus acinaciformis (L.) L. Bolus	2012	pe	f	INV	2.980	Reject	13
Carpobrotus edulis (L.) N.E. Br.	1981	pe	t	OBS	2.980	Reject	17
Casuarina equisetifolia L.	1964	pe	t	INV	2.601	Reject	8
Catalpa bignonioides Walter	2011	pe	s	MC	1.000	Accept	-13
Catharanthus roseus (L.) G. Don	2012	a	f	MC	1.020	Reject	9
Celtis occidentalis L.	2012	pe	g	MC	1.000	Accept	-3
Cenchrus longisetus M.C. Johnst.	1957	pe	t	INV	2.730	Reject	22
Chamaecyparis pisifera (Siebold & Zucc.) Endl.	1898	pe	h	MC	1.000	Accept	-12.5
Chamaesyce humifusa (Willd. ex Schltr.) Prokh.	2012	a	f	OBS	2.054	Reject	6
Chamaesyce maculata (L.) Small	1898	a	f	OBS	2.000	Reject	11
Chamaesyce prostrata (Aiton) Small	1964	pe	t	OBS	2.054	Reject	11
Chasmanthe aethiopica (L.) N.E. Br.	2000	a	f	MC	1.071	Accept	-3
Cicer arietinum L.	1894	a	f	MC	1.000	Accept	-9
Cichorium endivia L.	1844	a	f	MC	1.000	Accept	-8
Cinnamomum glanduliferum (Wall.) Meisn.	1957	pe	t	MC	1.000	Evaluate	2
Citrullus lanatus (Thunb.) Matsum. & Nakai	1808	a	f	MC	1.000	Accept	-8
Citrus limon (L.) Osbeck	1971	pe	s	MC	1.000	Accept	-3
Convolvulus tricolor L. subsp. tricolor	2008	pe	s	MC	1.000	Accept	-4
Cortaderia selloana (Schult. & Schult. f.)	2011	pe	ss	INV	2.990	Reject	11

Species	**First Record In TANP**	**Life Span**	**Life form**	**EPP list**	**EPP score**	**WRA list**	**WRA Score**
Asch. & Graebn.							
Cotoneaster coriaceus Franch.	2010	pe	g	OBS	2.094	Accept	-1
Cotyledon orbiculata L.	1966	pe	f	OBS	1.843	Accept	-5
Crassula lycopodioides Lam.	1898	pe	s	MC	1.000	Evaluate	3
Crataegus azarolus L.	2012	pe	ss	MC	1.000	Accept	-3
Cucurbita maxima Duchesne	1863	a	p	MC	1.000	Accept	-7
Cupressus sempervirens L.	2012	a	f	OBS	1.990	Accept	-7
Cuscuta epilinum Weihe	1888	pe	t	MC	1.319	Evaluate	2
Cycas revoluta Thunb.	2013	pe	g	MC	1.000	Accept	-11
Cydonia oblonga Mill.	1894	pe	t	MC	1.000	Accept	-9
Cyperus involucratus Rottb.	1974	pe	s	INV	2.875	Reject	8
Datura ferox L.	2012	a	f	OBS	1.973	Reject	23
Datura stramonium L. subsp. stramonium	1860	a	f	INV	2.812	Reject	12
Datura wrigthii Regel	2011	a	f	OBS	1.973	Reject	25
Delairea odorata Lem.	2001	pe	ss	INV	2.638	Reject	7
Delosperma cooperi (Hook. f.) L. Bolus	2010	pe	s	MC	1.000	Accept	-5
Diospyros kaki Thunb.	1964	pe	t	MC	1.000	Accept	-9
Diospyros lotus L.	1974	pe	t	MC	1.000	Accept	-9
Drosanthemum floribundum (Haw.) Schwantes	2011	pe	ss	MC	1.000	Reject	13
Dysphania ambrosioides (L.) Mosyakin & Clemants	2008	a	f	MC	1.340	Evaluate	1
Dysphania multifida (L.) Mosyakin & Clemants	1900	a	f	MC	1.340	Evaluate	2
Elaeagnus multiflora Thunb.	2012	a	g	MC	1.000	Accept	-3
Eleusine indica (L.) Gaertn. subsp. indica	1964	pe	s	INV	2.645	Reject	21
Ephedra foeminea Forssk.	1999	pe	ss	MC	1.000	Accept	-1
Erigeron bonariensis L.	1839	a	f	OBS	2.054	Reject	9
Erigeron canadensis L.	1839	a	f	OBS	2.054	Reject	9
Erigeron karvinskianus DC.	1844	pe	f	OBS	2.054	Reject	9
Erigeron sumatrensis Retz.	2012	pe	t	OBS	2.054	Reject	9
Eriobotrya japonica (Thunb.) Lindl.	1894	pe	s	MC	1.000	Accept	-10
Erysimum cheiri (L.) Crantz	1986	pe	f	MC	1.020	Accept	-4
Erythrostemon gilliesii (Wall. ex Hook.) Klotzsch	1972	a	f	MC	1.000	Accept	-3
Eucalyptus bicostata Maiden, Blakely & Simmonds	1975	pe	t	MC	1.000	Accept	-11
Eucalyptus camaldulensis Dehnh.	1964	pe	t	OBS	2.005	Reject	8
Eucalyptus cornuta Labill.	1964	pe	t	MC	1.000	Accept	-7
Eucalyptus globulus Labill.	1891	pe	t	OBS	2.005	Reject	6
Eucalyptus lehmannii (Schauer) Benth.	2011	pe	t	MC	1.000	Accept	-6
Euonymus japonicus Thunb.	1957	pe	t	MC	1.000	Accept	-3
Fallopia baldschuanica (Regel) Holub	2012	pe	s	OBS	2.005	Reject	18
Ficus elastica Roxb. ex Hornem.	1957	pe	t	MC	1.000	Accept	-8
Freesia alba (G.L. Mey.) Gumbl.	2012	pe	h	MC	1.000	Accept	-3
Gazania linearis (Thumb.) Druce	2012	pe	f	INV	2.792	Reject	8
Gleditsia triacanthos L.	2012	pe	t	INV	2.645	Reject	11
Grevillea robusta A. Cunn. ex R. Br.	1957	pe	t	MC	1.000	Evaluate	5
Helianthus annuus L.	1998	a	f	OBS	2.080	Evaluate	3
Helianthus tuberosus L.	1993	pe	f	INV	2.890	Reject	7
Hibiscus syriacus L.	1974	pe	s	MC	1.000	Accept	-3
Ipomoea indica (Burm.) Merr.	2008	pe	f	OBS	1.914	Reject	7
Iris germanica L.	1839	pe	h	MC	1.015	Accept	-5
Kalanchoë ×houghtonii D.B.Ward	2008	pe	ss	OBS	1.990	Reject	6
Lagerstroemia indica L.	2012	pe	s	MC	1.000	Evaluate	0
Lantana camara L.	1902	pe	s	INV	2.765	Reject	16

Species	First Record In TANP	Life Span	Life form	EPP list	EPP score	WRA list	WRA Score
Lens culinaris Medik.	1882	a	f	MC	1.055	Accept	-13
Lepidium didymum L.	2011	a	f	OBS	1.855	Evaluate	0
Ligustrum lucidum W.T. Aiton	1860	pe	h	INV	2.980	Reject	14
Lilium candidum L.	2008	pe	s	MC	1.000	Accept	-13
Linum usitatissimum L.	1893	a	f	MC	1.184	Evaluate	1
Lonicera japonica Thunb.	2012	pe	s	INV	2.875	Reject	6
Lupinus albus L. subsp. albus	1893	a	f	MC	1.350	Accept	-1
Lupinus polyphyllus Lindl.	1950	a	f	MC	1.000	Reject	7
Maclura pomifera (Raf.) C.K. Schneid.	1974	pe	t	OBS	1.997	Accept	-4
Magnolia grandiflora L.	1964	pe	t	MC	1.000	Accept	-4
Melia azedarach L.	2013	pe	t	INV	2.981	Reject	16
Mesembryanthemum cordifolium L. f.	1990	pe	ss	INV	2.980	Reject	10
Mirabilis jalapa L.	1993	pe	f	OBS	1.980	Evaluate	1
Morus alba L.	1836	pe	t	OBS	2.000	Evaluate	4
Morus nigra L.	1894	pe	t	OBS	2.000	Accept	-4
Myoporum insulare R. Br.	2011	pe	s	OBS	2.000	Accept	-8.5
Myoporum laetum Schltdl.	1964	pe	s	MC	1.000	Reject	9
Nicotiana glauca Graham	1976	pe	s	INV	2.800	Reject	12
Nolina recurvata (Lem.) Hemsl.	2012	pe	h	MC	1.000	Accept	-1
Nothoscordum gracile (Aiton) Stearn	1974	pe	s	MC	1.000	Evaluate	4
Ocimum basilicum L.	1900	a	f	MC	1.000	Accept	-8
Opuntia dillenii (Ker-Gawl.) Haw.	2003	pe	s	OBS	2.094	Reject	16
Opuntia elatior Mill.	2007	pe	s	MC	1.000	Accept	-7
Opuntia ficus-indica (L.) Mill.	1864	pe	s	INV	2.981	Reject	14
Opuntia leucotricha DC.	2013	pe	s	INV	2.980	Reject	9
Opuntia monacantha (Willd.) Haw.	1849	pe	s	INV	2.895	Reject	10
Opuntia phaeacantha Engelm.	2007	pe	s	INV	2.981	Reject	10
Opuntia stricta (Haw.) Haw.	1839	pe	s	INV	2.981	Reject	8
Osteospermum ecklonis (DC.) Norl.	2012	pe	ss	MC	1.000	Accept	-11
Oxalis articulata Savigny	1965	pe	h	OBS	2.000	Reject	9
Oxalis bowiei Herb. ex Lindl.	2012	pe	h	OBS	1.999	Reject	11
Oxalis debilis Kunth	2008	pe	h	OBS	2.020	Reject	9
Oxalis dillenii Jacq.	2000	pe	h	OBS	2.067	Reject	9
Oxalis latifolia Kunth	2013	pe	h	OBS	2.067	Reject	10
Oxalis pes-caprae L.	1955	pe	h	INV	2.990	Reject	9
Paraserianthes lophantha (Willd.) I.C. Nielsen	2012	pe	s	INV	2.792	Evaluate	5
Parthenocissus quinquefolia (L.) Planch.	2012	pe	g	INV	2.891	Reject	7
Parthenocissus tricuspidata (Siebold & Zucc.) Planch.	1947	pe	g	OBS	1.200	Accept	-8
Paspalum dilatatum Poir.	2009	pe	t	OBS	1.999	Reject	20
Paspalum distichum L.	2012	pe	s	INV	2.871	Reject	21
Passiflora caerulea L.	2012	pe	s	OBS	1.973	Evaluate	2
Pelargonium zonale (L.) L'Hér.	1957	pe	t	MC	1.000	Accept	-9
Persea americana Mill.	1839	b	f	MC	1.000	Accept	-7
Petroselinum crispum (Mill.) Fuss	2011	pe	ss	MC	1.000	Accept	-8
Phacelia tanacetifolia Benth.	1870	pe	f	OBS	1.997	Evaluate	4
Phoenix canariensis Chabaud	1981	pe	t	MC	1.000	Accept	-4
Phoenix dactylifera L.	1893	pe	t	MC	1.000	Accept	-4
Phytolacca americana L.	2011	a	f	OBS	2.020	Reject	19
Pinus canariensis C. Sm.	1950	pe	t	OBS	1.914	Evaluate	2
Pinus radiata D. Don	1974	pe	t	OBS	1.946	Accept	-4
Pisum sativum L. subsp. sativum	1839	a	f	MC	1.000	Accept	-16
Pittosporum tobira (Thunb.) W.T. Aiton	2013	pe	s	OBS	1.980	Reject	7
Platanus hispanica Mill. ex Münchh.	1981	pe	t	MC	1.000	Accept	-3
Platanus orientalis L.	1970	pe	t	MC	1.000	Evaluate	0

Species	First Record In TANP	Life Span	Life form	EPP list	EPP score	WRA list	WRA Score
Populus canadensis Moench	1964	pe	s	OBS	1.975	Reject	9
Portulacaria afra Jacq.	1993	pe	t	MC	1.000	Accept	-6
Prunus armeniaca L.	1898	pe	t	MC	1.000	Accept	-17
Prunus cerasifera Ehrh.	1974	pe	t	MC	1.000	Accept	-2
Prunus cerasus L.	1894	pe	t	MC	1.000	Accept	-5
Prunus domestica L. s.l.	1839	pe	t	MC	1.000	Accept	-1
Prunus persica (L.) Batsch	1894	pe	t	MC	1.000	Accept	-17
Pseudotsuga menziesii (Mirb.) Franco	1989	pe	t	MC	1.000	Accept	-4
Punica granatum L.	1894	pe	s	MC	1.000	Accept	-5
Pyrus communis L.	1839	pe	t	MC	1.000	Accept	-6
Quercus rubra L.	1957	pe	t	MC	1.000	Reject	6
Raphanus sativus L.	1898	b	f	OBS	1.949	Accept	-1
Robinia pseudoacacia L.	1974	pe	s	INV	2.980	Reject	14
Rosa multiflora Thunb.	1899	pe	t	MC	1.000	Reject	12
Salix babylonica L.	1900	pe	s	MC	1.000	Reject	13
Salvia officinalis L.	1833	pe	ss	MC	1.000	Accept	-7
Schinus molle L.	1957	pe	s	OBS	1.973	Reject	9
Senecio angulatus L. f.	1970	pe	s	INV	2.980	Reject	10
Setaria parviflora (Poir.) Kerguélen	2012	pe	g	OBS	2.030	Reject	6
Solanum lycopersicum L.	1976	a	g	MC	1.000	Accept	-10
Sophora japonica L.	1839	pe	g	MC	1.000	Accept	-4
Sorghum bicolor (L.) Moench	1964	pe	t	MC	1.000	Reject	6
Sorghum halepense (L.) Pers.	1839	a	f	INV	2.891	Reject	19
Sterculia diversifolia G. Don	1964	pe	s	MC	1.000	Accept	-4
Symphyotrichum squamatum (Spreng.) G.L. Nesom	1964	a	f	INV	2.808	Reject	10
Tamarix parviflora DC.	1894	pe	f	INV	2.601	Reject	9
Tanacetum parthenium (L.) Sch. Bip.	1972	pe	t	MC	1.000	Evaluate	2
Tetragonia tetragonoides (Pallas) Kuntze	2012	a	f	MC	1.000	Reject	6
Tilia tomentosa Moench	1977	pe	t	MC	1.000	Accept	-4
Tristania neriifolia (Sims) R. Br.	1900	a	g	MC	1.000	Accept	-2
Triticum aestivum L.	2008	a	f	MC	1.000	Accept	-1
Tropaeolum majus L.	1964	pe	s	OBS	2.000	Accept	-4
Tulipa clusiana DC.	1894	pe	h	MC	1.000	Accept	-4
Vachellia karroo (Hayne) Banfi & Galasso	2012	pe	t	INV	2.731	Reject	8
Veronica persica Poir.	1901	a	f	OBS	2.010	Evaluate	2
Vitis riparia Michx. x Vitis rupestris Scheele	2012	pe	s	INV	2.980	Evaluate	4
Xanthium orientale L. subsp. italicum (Moretti) Greuter	1870	a	f	INV	2.812	Reject	8
Xanthium spinosum L.	1864	a	f	INV	2.812	Reject	14
Yucca aloifolia L.	1898	pe	s	MC	1.000	Evaluate	0
Zantedeschia aethiopica (L.) Spreng.	2012	pe	h	INV	2.752	Reject	10
Ziziphus jujuba Mill.	1808	pe	s	OBS	2.139	Evaluate	4

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5.4 Predicting risk of invasion in a Mediterranean island using niche modelling and valuable biota.

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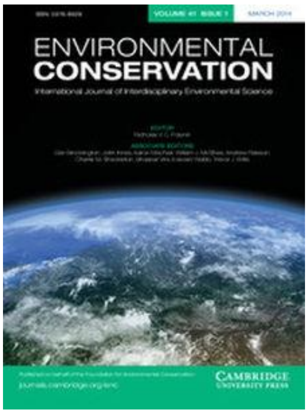
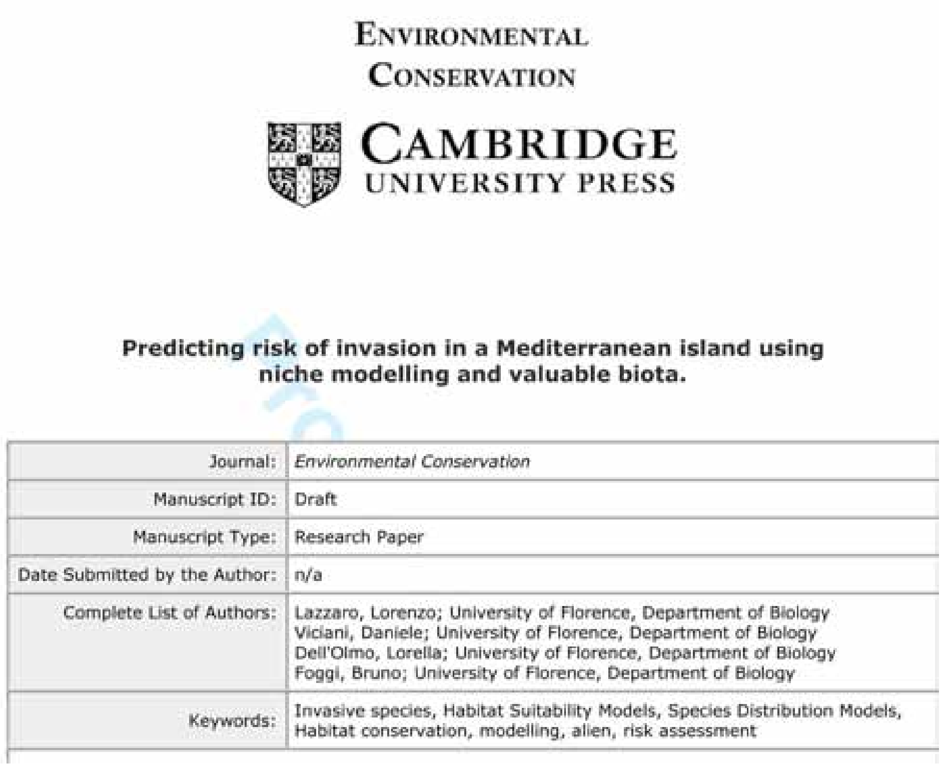
Submitted to *Environmental Conservation*





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SUMMARY

Invasion of ecosystems by alien species is nowadays considered one of the major threats to biodiversity. Effect on biodiversity are expected to be more dramatic on islands, due to their peculiar biome. In this context the identification of the areas exposed to a greater risk of invasion represents a priority for management purpose, especially in case they contain habitats worthy of conservation. This paper aims to propose a method to produce a map of risk of invasion, merging together the threat of invasion by invasive plants and the distribution of habitats with high conservation value. We illustrate this approach on the case study of a Mediterranean island (Island of Elba, Tuscan Archipelago). Towards this aim we modelled the potential distribution of six particularly harmful invasive species in Mediterranean ecosystems and merged these distributions into a map of threat of invasion. This map was overlapped to the map of density of Natura2000 habitats, finally obtaining the map of risk of invasion in the Island of Elba. According to our analyses the potential distribution of the invasive species resulted highly influenced by human related factors, such as the length of street per cell. The habitats main at risk are those closer to streets and anthropic habitats, which are more likely to be colonized by the invasive species we studied. We identified some rare habitats which are strongly endangered, highlighting that around 20% of the surface of the Island is exposed to some level of risk of invasion, determining an evident change in the landscape configuration.

KEYWORDS: Invasive species, Habitat Suitability Models, Species Distribution Models, Habitat conservation; modelling, alien, assessment.

INTRODUCTION

Nowadays biological invasions represent one of the major challenges in management and conservation efforts worldwide. Indeed the spread of Invasive Alien Species (IAS) is universally recognised as one of the greatest threats to the ecological and economic well-being of the planet (McNeely et al., 2001). Impacts of invasive alien plants are often related to diversity loss and richness decrease (Vilà *et al*. 2014; 2010; Pyšek *et al.* 2012), alteration of ecosystem function (Weidenhamer & Callaway 2010; Vilà *et al*. 2011), economic losses and relative control costs (Hulme *et al.* 2009; Scalera 2010) and to the globalization of local vegetation (Olden & Poff 2003).

It is widely accepted that biological invasion on islands are going to have even more dramatic effects due to islands' peculiar biota (Mayr 1967; Whittaker & Fernández-Palacios 2007). Islands host poor and disharmonious species assemblages, generally rich in endemics (Witthaker 1998) that may be particularly susceptible to plant invasion, i.e. with a possible change in species composition, with the replacement of endemic species with non-native ones (Vilà *et al*. 2014). Recently insularity and Mediterranean biome were depicted as important determinants of impacts on species richness (Pyšek *et al.* 2012).

Whereas prevention of further introduction, especially for known invasive species, represents the most effective management option, an essential strategy still consists in the constant monitoring of susceptible context and early response in case of invasion (McNeely et al. 2001). In this regard the identification of those areas that are more prone to be colonised represents a priority for management purpose, in order to optimise monitoring actions (Cronk & Fuller 1995; Macdonald 1990), especially in case they contain habitats worthy of conservation. One of the main issues for land managers and nature conservationist is to focus conservation efforts and resources on priority contexts. Indeed the assessment of risk of invasion should take in account two main components: the likelihood of invasion and the potential presence of impacts (Gassó et al. 2009 and references therein). Particularly to risk assessment in biological invasions, it is noteworthy that the major impacts of invasive species are considered a priority when exerted on natural and valuable biotas (see for example EPPO 2012; D'Hondt et al. 2014).

In this framework Species Distribution Models (SDMs) represent an essential tool allowing to predict the potential geographical distribution of alien species (Peterson 2003, Jarnevich *et al.* 2010) and can be used by stakeholders to prioritise monitoring efforts (Crall *et al.* 2012 and reference therein). SDMs are useful tools for resolving practical questions in applied ecology and conservation biology (Guisan & Thuillier 2005) and play a very important role in the development of conservation planning (McShea 2014). SDM concept is to create a predicted (or potential) distribution of one or more species extrapolate from known distribution of species and a set of critical environmental variables (Guisan & Zimmermann 2000, McShea, 2014).

Habitat is considered a central pillar of nature conservation policy, and the maintenance of a series of habitats in good condition is one of the best ways to conserve species (Bunce *et al.* 2013). Furthermore habitats offer a great opportunity in conservation. They can indeed be considered as real operational objects, expressed in the form of spatially recognisable patches, having a distinct aspect that makes them perceivable as land elements (‘patches’) or, at least, convenience units (Wilson & Chiarucci 2000). Habitat concept is largely acknowledged in the scientific community and have an essential role in the European legislation, thanks to the European Habitats Directive (92/43/EEC) which relied on the phytosociological approach for identifying habitats deserving the highest effort of conservation (Rodwell *et al.* 2002; Evans 2006; 2010; Biondi *et al.* 2012). The Directive provided an agreed list of habitat types (Annex I, European Commission 2013) to be preserved in the Natura2000’s European network.

Modelling the potential presence of invasive species, and assess where these species may affect valuable biotas, may represent a useful and practical tool for management purposes. In this perspective, habitats of conservation concern, according to the Directive 92/43, can be used as “proxy” of areas were the invasion can determine massive consequences.

Aim of this paper was to propose a method to produce a map of risk of invasion, merging together the threat of invasion by invasive plants and the distribution of habitats with high conservation value, highlighting where the potential distribution of IAS threaten the areas with high conservation values. Toward these aims we 1) produced a map of the threat of invasion modelling the potential distribution of six well known invasive plants in a Mediterranean island 2) compared the threat of invasion to the valuable biota, expressed as density of Natura2000 habitats and 3) assessed the risk of invasion on the area of study.

These products will enable the Local Administrator to prioritize the management costs in the higher risk contexts for the application of focused measures of control and prevention on a specific area.

METHODS

Study Area

The Island of Elba is the main island of the Tuscan Archipelago, and the third of Italy (with 223 square kilometres of land surface). It is located between 42° 40' and 42° 55' N and 10° and 10° 30' E, in Central Mediterranean, west of Tuscany, Italy. More than half of the island surface (about 128 square kilometres) is part of the Tuscan Archipelago National Park. Resident population is of 31342 inhabitants (data ISTAT 2011), but the island is interested by intense touristic flow. Indeed in the last 50 years the island was involved in a transition from an economy based largely on agricultural exploitation to one based on tourism development, also determining a substantial shift in the land uses.

The orography of the island is quite heterogeneous, ranging from about 1050 meters of the Capanne Mt. to sea level, and it is mainly represented by medium-low altitude hills. The climate is typically Mediterranean, with a primary maximum of precipitation in autumn, a second maximum in winter and a main minimum in summer (Maselli *et al.* 2000). Mean annual temperature ranges from 10°C at the maximum elevation to 17°C at the sea level. Mean temperature ranges from 21.1°C to 24.3°C in the warmest month (July) and 2.4°C to 10.6°C in the coldest month (January). The mean annual rainfall is 763.2 mm/year, but is strictly related to the altitude and ranges from 550 mm at the sea level to 1372 mm/year at the higher elevations (Foggi *et al.* 2006). Soils are prevalently xeric, characterized by a scarce water retention capacity during the recharging period and high evapotranspiration during the summer (Maselli *et al.* 2000).

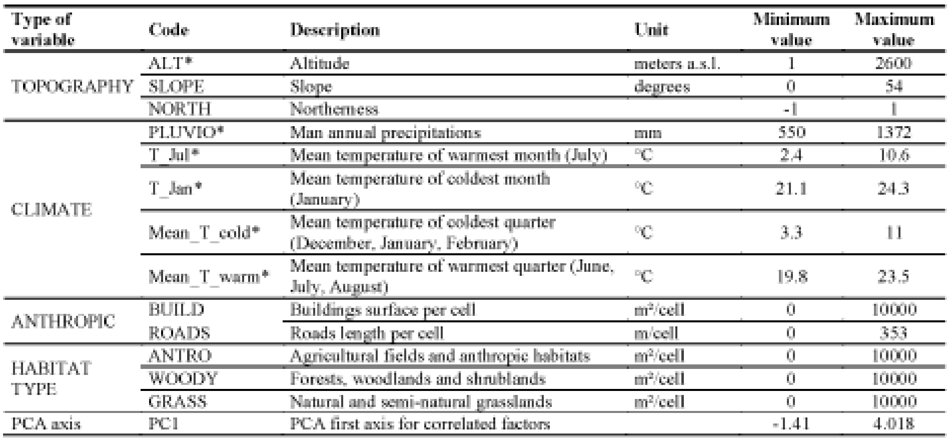
The landscape is dominated by a typical Mediterranean sclerophyllous–evergreen forest and by its degradation stages, such as high and low matorrals, garrigues and discontinuous ephemeral grasslands (Foggi *et al*. 2006). Natural and semi-natural habitats cover almost the 80% of the territory, with the remaining surface occupied by artificial (11%) and agricultural lands (9%). According to the European Habitat Directive (EC 1992; 2013), the island hosts 27 different habitat types of conservation interest, covering the 39% of the surface of the island (Viciani *et al.* in press), and several plant species of conservation relevance according to Tuscan laws and projects (http://www502.regione.toscana.it/geoscopio/arprot.html). Indeed, the Island of Elba is one of the areas with the highest concentration of both habitat reports and habitat types per unit area in Tuscany (Viciani *et al.* in press). All habitats of conservation interest, according to the 92/43 Habitat Directive are reported in Table S1 (Appendix1, Table S1, see supplementary material at Journals.cambridge.org/ENC).

Invasive Aliens Species distribution data

From those species which shown invasive behaviour in the Island of Elba (Lazzaro *et al.* 2014) and are widely known as invasive species in Mediterranean ecosystems (Weber *et al.* 2003; Gassó *et al*. 2009; Brundu 2013; Foggi *et al.* 2014), we selected six species for which an appropriate quantity of distributional information was available. These species can be considered some of the worst found in the Tuscan Archipelago, according to the assessment produced with the EPPO (EPPO 2012) method (Lazzaro *et al.* 2013). The selected species are *Acacia dealbata*, *Agave americana*, *Ailanthus altissima*, *Opuntia ficus-indica*, *Oxalis pes-caprae* and *Robinia pseudoacacia*. We gathered presence/absence information on these from two main sources. We firstly selected all presence data coming from local literature and reports which were geographically accurate (which spatial location could be retrieved with a 20 m accuracy). Thus, we improved this dataset with an intensive field survey carried out across 2012 and 2013. In this case presence data were collected exploiting a GPS device. Finally we resampled the presence data for each species on a 100m x100m regular grid, resolution at which also all the environmental variables were resampled and at which we conduced all the analyses. This lead to a grid of 23221 cells. The number of occurrences of the alien species ranged from 95 for *Ailanthus altissima* to 150 for *Acacia dealbata* (see Appendix1, Table S2 for full taxonomy, family and number of occurrence for species. See supplementary material at Journals.cambridge.org/ENC).

Environmental data

We chose four types of predictors from climate, topography, anthropogenic factors and land-use, to describe the environmental space of the species. The climate variables included average annual precipitation in mm and annual average temperature, average temperature of the warmest month, average temperature of the coldest month and average temperature of colder and warmest quarters (see tab. 1). Across climate variable these are considered very important in predicting plant species distribution (Rivas-Martínez & Rivas-Sáenz 2009). Climate variables were extracted from Foggi *et al.* (2006) and resampled at the adopted grid. Topographical factors were elevation, slope, and slope aspect (transformed into Northerness according to the formula: Northerness = cosine [(aspect in degrees * π)/180)].



Type of variable	Code	Description	Unit	Minimum value	Maximum value
TOPOGRAPHY	ALT*	Altitude	meters a.s.l.	1	2600
	SLOPE	Slope	degrees	0	54
	NORTH	Northerness		-1	1
CLIMATE	PLUVIO*	Man annual precipitations	mm	550	1372
	T_Jul*	Mean temperature of warmest month (July)	°C	2.4	10.6
	T_Jan*	Mean temperature of coldest month (January)	°C	21.1	24.3
	Mean_T_cold*	Mean temperature of coldest quarter (December, January, February)	°C	3.3	11
	Mean_T_warm*	Mean temperature of warmest quarter (June, July, August)	°C	19.8	23.5
ANTHROPIC	BUILD	Buildings surface per cell	m²/cell	0	10000
	ROADS	Roads length per cell	m/cell	0	353
HABITAT TYPE	ANTRO	Agricultural fields and anthropic habitats	m²/cell	0	10000
	WOODY	Forests, woodlands and shrublands	m²/cell	0	10000
	GRASS	Natural and semi-natural grasslands	m²/cell	0	10000
PCA axis	PC1	PCA first axis for correlated factors		-1.41	4.018

Table 1 Summary of the explanatory variables exploited for the analysis. Variables marked with a * were excluded from the models because their information was summarized by PCA first axis.

Topographic factors were derived directly from the Digital Elevation Model (DEM). Anthropogenic factors were extracted from the CTR cartography (available at http://www502.regione.toscana.it/geoscopio/cartoteca.html). From this cartography we extracted the length of street per cell and the surface of buildings per cell. To include the land-use type, which can embrace many factors, we used physiognomic vegetation types and artificial land-use typologies, extracted from the vegetation map of Island of Elba (Foggi *et al.* 2006). We distinguished between human related land-use types, including agricultural fields and urban areas; woody habitats, including all natural and semi-natural forests, woodlands and shrublands and grasslands, including all permanent grasslands. The predictive factors adopted in the models are shown in Table 1.

We tested the correlation between environmental variables exploiting the Pearson's correlation coefficients (Rp). Particularly topographic (altitude) and all the climatic variables showed to be highly correlated variables (Rp> |0.7|). This can be expected giving the size of the area of study, where the main driver of climatic variation is represented by altitude. To remove redundancy among correlated variables we performed a Principal Components Analysis (PCA). Thus, adopting the "broken-stick" rule, which is considered to work quite well for highly correlated variables (Peres-Neto *et al.* 2005), we kept only the first PCA axis to summarize the variation related to these variables.

All the spatial analyses have been carried out using the ESRI ArcGIS Desktop 9.2 software. Correlation analysis has been carried out exploiting the *stats* package and ordination analysis using the *vegan* package version 2.0-7 (Oksanen *et al*. 2013) of R Software version 3.1.2 (R Core Team 2014).

Ecological niche modelling

The ecological niche modelling has been carried out exploiting the R package *biomod2* version 3.1-48 by Thullier *et al* (2014). This package allows to run 10 state-of-the-art modelling techniques to predict species distribution in function of environmental explanatory variables (Thullier *et al* 2014). According to Barbet-Massin *et al.* (2012) we chose to run all the models included in the biomod2 package. To avoid problems related to overdispersion of the data caused by the huge amount of absence of the species compared to presence and to avoid the misuse of a lack of information being interpreted as absence of the species, we exploited pseudo-absences (PA) instead of true absences to fit the models. According to Barbet-Massin *et al.* (2012), aiming to obtain a PA selection which would be good for all the algorithm adopted, we used 10 set of PA in equal number of species presence random selected. As commonly done (Philips *et al.* 2006) we chose to use the 70% of occurrence records of each species to calibrate the model and 30 % to test it. We set as default all the other parameter in the different model options. The performances of the models were assessed using the Area Under the Curve (AUC) of Receiver Operating Characteristic (ROC) curves (Hanley & McNeil 1982).

Then for each species we produced a total consensus model including all models with AUC > 0.85, using the mean of probabilities of selected models as modelling algorithm. The performance of single species total consensus models were again assessed using AUC. Finally we derived the potential presence from the continuous logistic output using the cut-off threshold suggested by ROC analysis.The model procedure was repeated in a loop for each of the six species to finally produce six map of potential distribution for the selected IAS.

Risk of invasion on the Island of Elba

Our Risk map approach started from the classical theory of Risk evaluation, were potential Risk is given by Threat * Vulnerability. In our case threat is represented by the potential invasion by IAS and the vulnerability is represented by valuable biota present in the cell.

To create a “Map of Threat of invasion for the Island of Elba” we merged together the six maps of potential presence for the six IAS. The presence/absence values (1/0) of each species were added together for each cell of the grid using the following formula:

$$TP = \sum p_i$$

where TP is the potential threat of invasion and p_i represents the potential presence/absence value of *i*-esim species in the cell. TP should be interpreted as the number of invasive species that may colonize the cell and thus varies from 0, in case of no potential presence falling in the cell, to 6 in case of the potential presence of all the invasive species in the cell.

A “Map of the density of Natura2000 habitats” was designed to express the presence of valuable biota in the Island of Elba (Appendix 1, Figure S3, See supplementary material at Journals.cambridge.org/ENC). The map is derived from the map of Natura2000 habitats of the Island of Elba as presented in Viciani et al., (in press). The habitat 1160 was excluded from the elaboration being a submerged aquatic habitat, thus unlikely threatened by terrestrial invasive plant. The density of Natura2000 habitats is given at the same resolution adopted for the maps of potential distribution of IAS in the Island of Elba, calculated as the number of habitat types that can be found in each cell. This number was obtained with a series of queries and spatial joins in ArcGIS.

Finally the Map of Threat of invasion for the Island of Elba and the Map of the density of Natura2000 habitats were overlapped following the formula:

$$R_{inv}(Hab) = TP * Hd$$

where the Risk of invasion of the habitats, $R_{inv}(Hab)$, is calculated as the product of the potential threat of invasion (TP) times the Natura2000 habitat density (Hd). The value of $R_{inv}(Hab)$ reflects the net number of habitat times the number of IAS potential presence in the cell and is therefore of difficult interpretation. Thus the value was ranked in three classes (low, medium and high risk, and excluding the 0 considered as absence of risk), using the Natural Breaks (Jenks) method provided in ArcGIS as a classification choice.

RESULTS

Ecological niche modelling

A total of 600 models were run, 100 for each species (10 modelling algorithms per 10 Pseudo Absence set repetition). Table 2 summarise the performances of the models according to the AUC values. These were ranging from 0.419 to 0.986, with a mean of 0.858 ± 0.105 (SD). Generally the mean AUC value for each species was above 0.80. The number of models which passed the threshold of AUC>0.85 was different among the species. *A. dealbata* had the lowest number of accepted models (33 out of 100), whereas for the other species these were around 60-80 % of the total (Table 2). AUC values of the ensemble models were quite good, ranging from 0.930 to 0.960 (Table 3). Generally all models showed better performances in term of sensitivity than specificity.

Species	Mean AUC Value	AUC values SD	Min AUC value	Max AUC value	Number of models with AUC value > 0.85	Number of models with AUC value > 0.90
Acacia dealbata	0.802	0.108	0.467	0.965	33	10
Agave americana	0.850	0.096	0.468	0.967	64	27
Ailanthus altissima	0.868	0.084	0.517	0.970	74	38
Opuntia ficus-indica	0.861	0.108	0.478	0.965	78	45
Oxalis pes-caparae	0.889	0.091	0.559	0.976	84	61
Robinia pseudoacacia	0.877	0.121	0.419	0.986	82	59
TOT	0.858	0.105	0.419	0.986	415	240

Table 2 Summary of AUC values of Receiver Operating Characteristic (ROC) curves obtained for the single species models. AUC = Area Under the Curve. SD = Standard Deviation

Species	AUC	Cut-off threshold	Sensitivity	Specificity
Acacia dealbata	0.933	492.5	92.667	80.989
Agave americana	0.943	338.5	95.146	83.342
Ailanthus altissima	0.960	211.0	98.947	83.300
Opuntia ficus-indica	0.948	322.5	95.302	81.597
Oxalis pes-caparae	0.957	370.5	97.368	83.901
Robinia pseudoacacia	0.960	208.5	99.187	82.540

Table 3 Performance and characteristics of the single species total consensus models. AUC = Area under the Curve of Receiver Operating Characteristic (ROC) curves. Cut-off threshold = logistic value above which potential presence are estimated. Sensitivity represents the proportion of true positives. Specificity represents the proportion of true negatives.

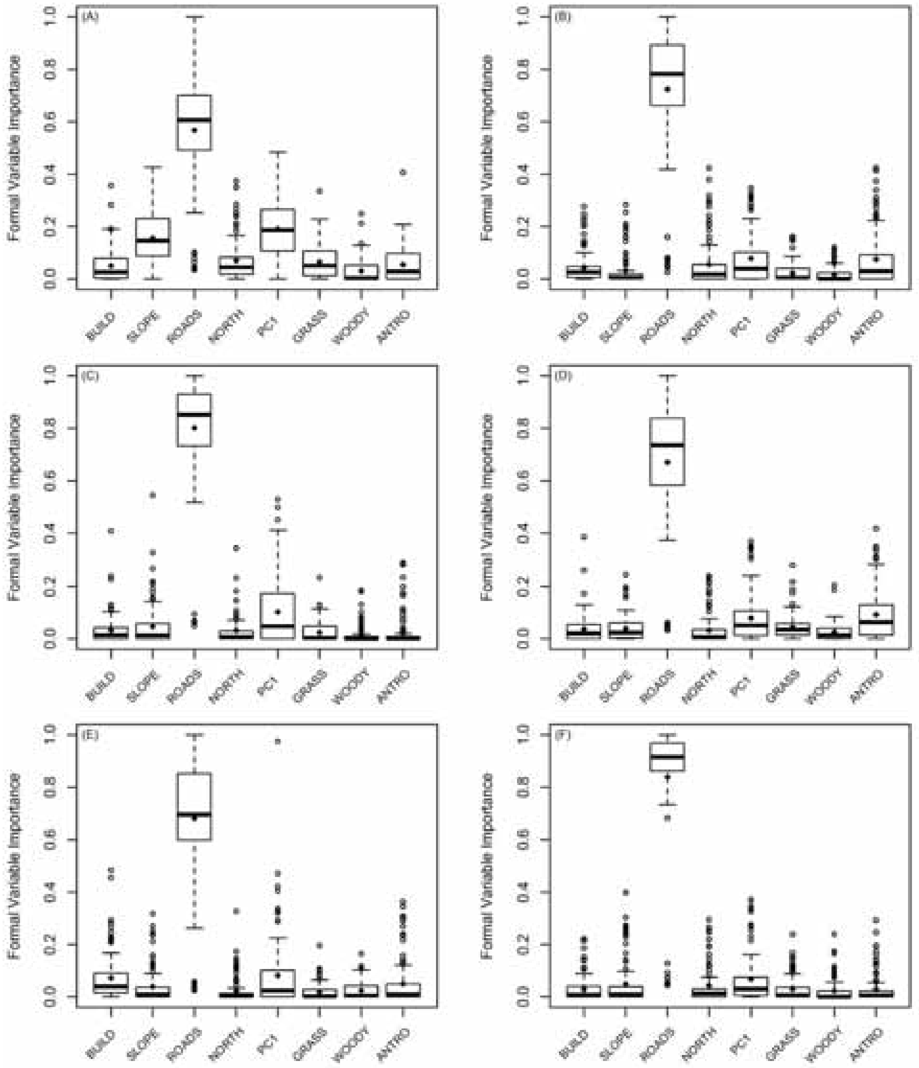


Figure 1 Box plot diagrams for formal importance of predictive variables adopted in the models. Species: (A): *Acacia dealbata*; (B): *Agave americana*; (C): *Ailanthus altissima*; (D): *Opuntia ficus-indica*; (E): *Oxalis pes-caprae* and (F): *Robinia pseudoacacia*. Bold line: median, rhombus: mean, box: interquartile range, whiskers: typical range, points: outliers.

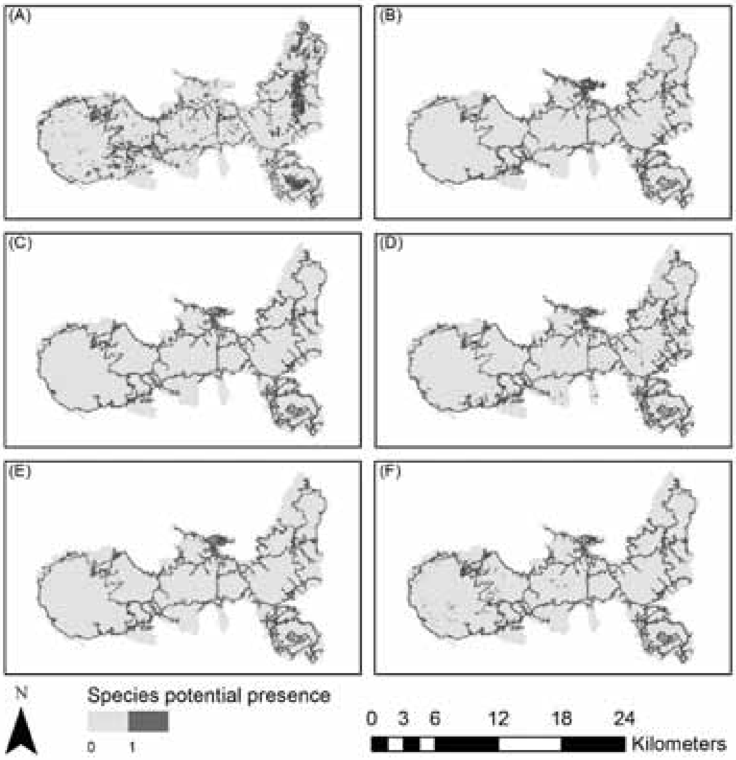


Figure 2 Maps of potential presence for the six invasive alien plants. Species: (A): *Acacia dealbata*; (B): *Agave americana*; (C): *Ailanthus altissima*; (D): *Opuntia ficus-indica*; (E): *Oxalis pes-caprae* and (F): *Robinia pseudoacacia*.

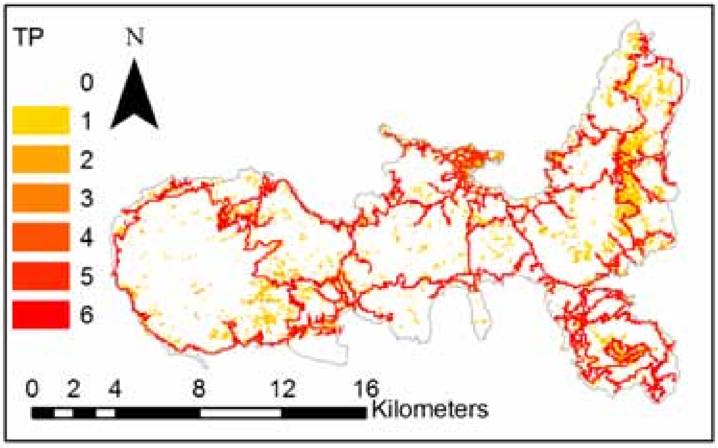


Figure 3 Map of Threat of Invasion for the Island of Elba. TP = potential threat of invasion obtained as the sum of the potential presences of the six invasive species.

The analyses of formal importance of the variables (Fig. 1) showed quite clearly that length of roads per cell is generally the most important factor determining the presence of the species. While the importance the other factors differ among the species. PC1, summarizing altitude and climatic variables, showed to be important for few species, mainly Acacia dealbata and Ailanthus altissima.

The generally high importance of length of roads per cell is reflected in the resulting maps of potential presence for the six invasive species (Fig. 2) and even more in the Map of Threat of invasion for the Island of Elba (Fig. 3).

Risk of invasion on the Island of Elba

The map on (Fig. 4) represent the Risk of invasion in the Island of Elba. Around 80% of the cells have a value of risk equal to 0, being cells without any potential IAS presence or without valuable habitats. The remaining 20% of cells were assigned according to Jenks Natural Breaks in the three classes of low risk (1700 cells, 7.32% of the total), medium risk (1858 cells, 8.00% of the total) and high risk (928 cells, 4.00% of the total) (Table 4).

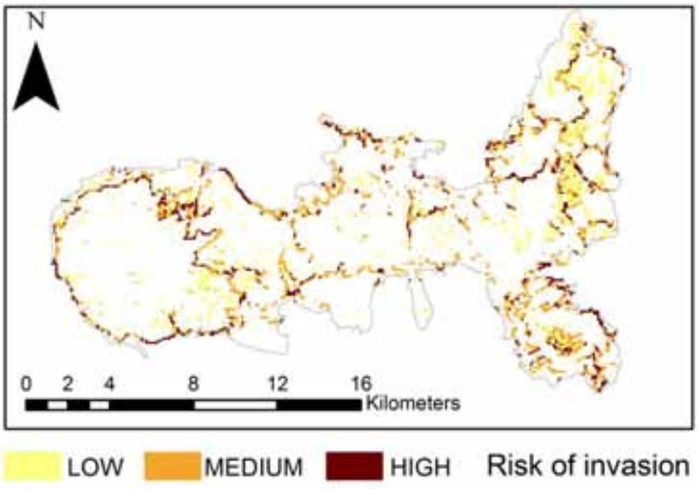


Figure 4 Map of Risk of Invasion in the Island of Elba.

The GIS database obtained during the map processing is quite deeply queryable (Appendix 2, See supplementary material at Journals.cambridge.org/ENC). In fact for each cell, in addition to the risk value, it is possible to retrieve the information that produced this value. Thus it is possible to extract the number of potential IAS presence on the cell (and also the identity of IAS contributing to that value) and the density of habitats (and deeper again, the specific habitat types contained in the cell). Accordingly it was possible to calculate and analyse the distribution in the three classes of risk of each habitat (Appendix 1, Figure S4, See supplementary material at Journals.cambridge.org/ENC). Fig. 5 shows the first 13 habitats most at risk. These habitats are exposed to some degree of risk of invasion in more than the 30% of the cells in which they are present. Furthermore, most of these are quite rare habitats, while only three of them are present in a high number of cells.

Net Risk Value	Number of cells	Proportion on total cells (%)	Risk class
0	18735	80.68%	NONE: 18735 cells (80.68%)
1	888	3.82%	LOW: 1701 cells (7.32%)
2	643	2.77%	
3	169	0.73%	
4	172	0.74%	MEDIUM: 1857 cells (8.00%)
5	220	0.95%	
6	1422	6.12%	
7	1	0.00%	
8	35	0.15%	
9	8	0.03%	
10	96	0.41%	HIGH: 928 cells (4.00%)
12	578	2.49%	
15	37	0.16%	
18	151	0.65%	
20	16	0.07%	
24	46	0.20%	
25	1	0.00%	
30	3	0.01%	

Table 4 Summary of Value of Risk of invasion per cell. Net Risk Value is the product of number of habitat times the number of IAS potential presence in the cell. Classes of risk were obtained using the Natural Breaks (Jenks) method provided in ArcGIS software.

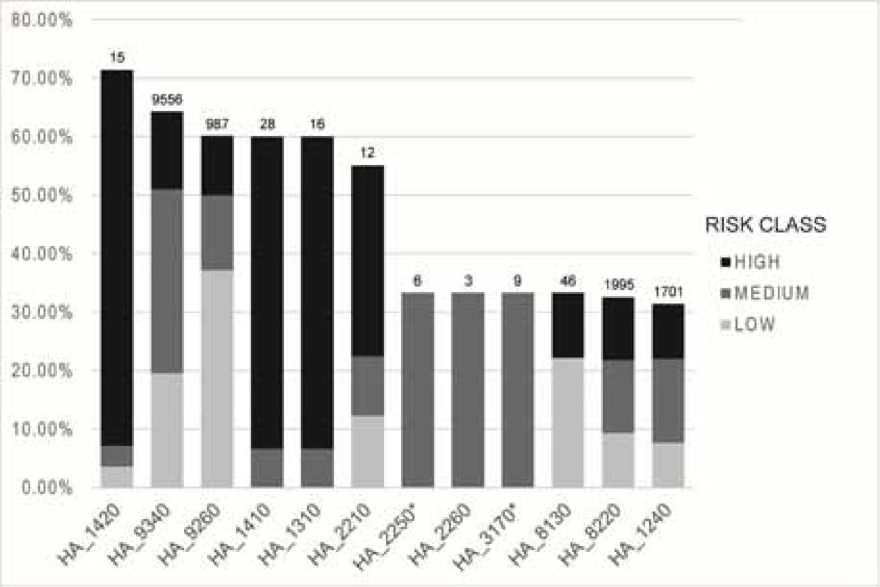


Figure 5 Level of risk for habitat's type. Habitat codes according 92/43 Directive, particularly first digit refers to: 1 and 2 = coastal habitats, 3 = aquatic habitats, 8 = rocky habitats, 9 = forest habitats. For further details see Appendix 2. Numbers above each bars indicate the total number of cells in which the habitat can be found on the Island of Elba.

DISCUSSION

Aim of this work was to evaluate the risk of invasion of valuable biota on the Island of Elba overlapping the potential threat of invasion with the presence of valuable biota. We used SDMs of six particularly harmful species to assess the threat of invasion and the Natura2000 habitats, highlighting that around 20% of the island is exposed to some degree of risk of invasion.

SDMs represent a valuable tool in conservation biology, and especially in last decades they have been largely used (Peterson 2007). However some drawbacks in their use to assess the level of threat of invasion have been largely discussed (Jiménez-Valverde *et al.* 2008; Hortal *et al.* 2010). These pitfalls mainly regard the fact that species may not be in the equilibrium with the environment and, especially regarding alien species, may still not to occupy all suitable places in the area under study (Peterson 2005). This would lead to an underestimation of the suitable habitat in spite of the goodness of fit of the models (Jiménez-Valverde *et al.* 2008). One more issue is related to the reliability of data adopted for the modelization, consequently influencing the reliability of the results of the modelling process (Hortal *et al*. 2010). Even more in case of alien species when only occurrence data from the invaded area are used in the modelling process (Mau-Crimmins *et al.* 2006). Acknowledging these pitfalls, SDMs still represent a quite useful tool when is adopted a suitable treatment of the available data. In our case we started with an *ad hoc* field survey aiming to produce an adequate set of distributional data underlying the modelling process. We adopted several replicates of pseudo absences selections aiming to minimize the underestimation due to lack of equilibrium of species with the environment.

Following the classification proposed by Swets (1988)(AUC values: 0.90-1.00 = excellent; 0.80-0.90 = good; 0.70-0.80 = fair; 0.60-0.70 = poor; 0.50-0.60 = fail), the fitted models resulted generally "good". Mean AUC values were always above 0.80 and around 70% of the models passed the threshold of AUC >0.85. This is reflected in the production of "excellent" ensemble models.

The analysis of relative importance of factors in the modelling process highlighted the essential role of human presence (i.e. presence of streets) in determining the presence of IAS in the area of study. The essential role of roads and streets in the spread of IAS have been largely demonstrated (Gelbard & Belnap 2003; Hansen & Clevengen 2005; Douglas *et al.* 2009). Particularly roads function as conduits for the spread of IAS through two main mechanisms, representing: 1) an essential way of dispersal for non-native species and 2) a simplified and human disturbed suitable habitat which have an extremely high permeability to invasion (Douglas *et al.* 2009). Furthermore, specifically to Mediterranean islands, Pretto *et al* (2013) have already highlighted the predominant role played by the artificial surfaces on the richness and composition of non-native flora on small Mediterranean islands. Indeed already Vilà *et al* (2008) suggested that alien plants tend to settle primarily in areas heavily subject to human disturbance, while natural and semi-natural areas showed a certain resistance to the invasion. In our case anthropic land uses and presence of buildings are the second factor (i.e. see *Opuntia ficus-indica* and *Agave americana*) favouring invasions. Thus, also seen that actually most of the species appear distributed along roads and on anthropic habitats, it is convincing that areas closer to such habitats will be the first reached by these alien plants. Minor importance of climatic factor (summarized in PC1) is well explained by the general uniformity of climatic conditions in the area of study, where the main source of climatic variability is represented by the altitude gradient along the slopes of Mount Capanne, in the Western part of the Island of Elba.

IAS are widely known as a harmful threat to biodiversity, especially at local scale (Vilà *et al.* 2014). For instance according to Viciani *et al* (2014) IAS represent the second cause of impact to Tuscan habitats, between natural and semi-natural threats and pressures. This appear even more important considering an island ecosystem (De Montmollin & Strahm 2005), such as the Island of Elba.

Among the habitats the risk of invasion was quite heterogeneous. Some habitats are quite abundant in the area of study, and thus also quite at risk. On the other hand there are some other habitats pretty rare and highly endangered by risk of invasion.

Coastal habitats are ones of most threatened, especially those of the low coast brackish environments (e.g.: 1310: *Salicornia* and other annuals colonizing mud and sand; 1410: Mediterranean salt meadows; 1420: Mediterranean and thermo-Atlantic halophilous scrubs) and sand dunes (e.g.: 2210: *Crucianellion maritimae* fixed beach dunes; 2250: Coastal dunes with *Juniperus* spp.). It is noteworthy that the latter are relict habitats in the Island of Elba, nowadays present only in small areas subjected to intense tourism exploitation and consequently affected by urbanization and road network.

Also the rocky habitats of coasts (i.e. 1240: vegetated sea cliffs of the Mediterranean coasts with endemic *Limonium* spp.) and inlands (i.e. 8130: Western Mediterranean and thermophilous scree; 8220: siliceous rocky slopes with chasmophytic vegetation), although resulting generally less sensitive, are subject to high risk of invasion for a considerable proportion of locations. This is because some of the invasive species here tested (i.e. *O. ficus-indica* and *A. americana*) become more aggressive in these environments. Among the forest habitats, those that are fairly widespread in the island of Elba (i.e. 9340: *Quercus ilex* forests; 9260: *Castanea sativa* woods) are subject to a significant risk. This is particularly important since these communities appear floristically different from the analogous ones of the Italian peninsula (Foggi *et al.* 2006.

A procedure of risk assessment based on habitats appears of great importance in the light of the recent trend to consider more valuable the use as the unit of assessment of vulnerability a category of higher order than species (such as ecological communities, habitats, etc.) (Nicholson *et al.* 2009; Rodriguez *et al.* 2011; 2012). In fact, recently, a specific focus on the assessment of vulnerability of habitats has been developed by European Union, in the perspective of the establishment of a European Red List of habitats (Rodwell *et al.* 2013). Furthermore also the International Union for Conservation of Nature (IUCN) is developing an assessment procedure according to a standards similar to the one used for the species according to Keith *et al.* (2013). In this perspective, risk of invasion of a given habitat in a given area should represent a valuable additional criterion to be taken into account in the evaluation criteria.

In conclusion, our work offers a practical tool adopting IAS SDMs and valuable biota toward the improvement of risk management options. The six invasive species SDMs allow to assess the potential risk of invasion of valuable habitats on the Island of Elba. According to our analyses the potential distribution of the invasive species resulted highly influenced by human related factors, such as the length of streets per cell. Thanks to our procedure we individuated the habitats which represent a priorities in management options and the areas where these are going to be potentially endangered. The habitats main at risk are those closer to streets and anthropic habitats, which are more likely to be colonized by the invasive species we studied. We identified some rare habitats which are strongly endangered, highlighting that around 20% of the surface of the Island is exposed to some level of risk of invasion, determining an evident change in the landscape configuration. Specific policies and strategies, also for management at a local level, are urgently needed focusing on the individuation of priorities of intervention and the monitoring of key cases.

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Supplementary material – Appemdix 1

Natura2000 habitat code	Name	Surface (Hectares)	Proportion on the total habitats surface (%)
	Coastal and Halophytic Habitats		
1160	Large shallow inlets and bays	4.2	0.048
1210	Annual vegetation of drift lines	0.2	<0.1
1240	Vegetated sea cliffs of the Mediterraneancoasts with endemic *Limonium* spp.	174.3	1.997
1310	*Salicornia* and other annuals colonising mud and sand	0.3	<0.1
1410	Mediterranean salt meadows (*Juncetalia maritimi*)	1.3	0.015
1420	Mediterranean and thermo-Atlantic halophilous scrubs (*Sarcocornetea fruticosi*)	0.8	0.009
	Coastal Sand Dunes		
2110	Embryonic shifting dunes	0.4	<0.1
2210	*Crucianellion maritimae* fixed beach dunes	0.2	<0.1
2230	*Malcolmietalia* dune grasslands	0.1	<0.1
2250*	Coastal dunes with *Juniperus* spp.	0.7	<0.1
2260	*Cisto-Lavanduletalia* dune sclerophyllous scrubs	0.5	<0.1
	Freshwaters Habitats		
3120	Dwarf amphibious vegetation of oligotrophic waters of the West Mediterranean with *Isoetes* spp.	<0.1	<0.1
3170*	Mediterranean temporary ponds	<0.1	<0.1
	Temperate Heath and Scrub		
4090	Endemic oro-Mediterranean heaths with gorse	166.2	1.903
	Sclerophyllous Scrub		
5210	Arborescent matorral with *Juniperus* spp.	30.6	0.350
5320	Low formations of *Euphorbia* close to cliffs	116.2	1.331
5330	Thermo-Mediterranean and pre-desert scrub	379.6	4.349
	Natural and Seminatural Grasslands		
6220*	Pseudo-steppe with grasses and annuals of the *Thero-Brachypodietea*	319.5	3.660
6420	Mediterranean tall humid herb grasslands of the *Molinio-Holoschoenion*	42.3	0.485
	Rocky Habitats		
8130	Western Mediterranean and thermophilous scree	9.2	0.105
8220	Siliceous rocky slopes with chasmophytic vegetation	349.7	4.005
	Forests		
91E0*	Alluvial forests with *Alnus glutinosa* and *Fraxinus excelsior*	75.2	0.862
91F0	Riparian mixed forests of *Quercus robur, Ulmus laevis* and *U. minor, Fraxinus excelsior* or *F. angustifolia*	5.3	0.061
9260	*Castanea sativa* woods	400.0	4.582
9330	*Quercus suber* forests	288.7	3.308
9340	*Quercus ilex* and *Quercus rotundifolia* forests	5009.9	57.389
9540	Mediterranean pine forests with endemic Mesogean pines	1354.2	15.512
TOTAL		8729.8	100.000

Table S1. List of Natura2000 Habitats found on the Island of Elba, with code and full name according Natura2000, arranged for principal type of ecosystem, distribution area in Elba expressed in hectares and in percentage. * marks priority habitat according European Habitat Directive (EC 1992; 2013).

Specie	Family	Number of occurrences
Acacia dealbata Link	Fabaceae	150
Agave americana L.	Asparagaceae	103
Ailanthus altissima (Mill.)Swingle	Simarubaceae	95
Opuntia ficus-indica (L.) Mill.	Cactaceae	149
Oxalis pes-caprae L.	Oxalidaceae	114
Robinia pseudoacacia L.	Fabaceae	123

Table S2 Full taxonomy, family and number of occurrence for species for the invasive alien species selected for the models.

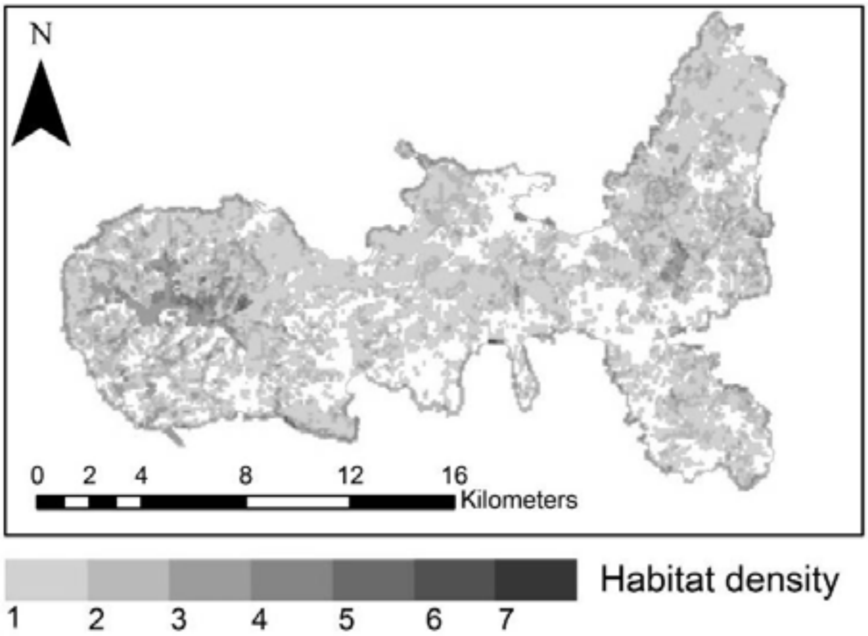

Figure S3 Map of the density of Natura2000 habitats for the Island of Elba.

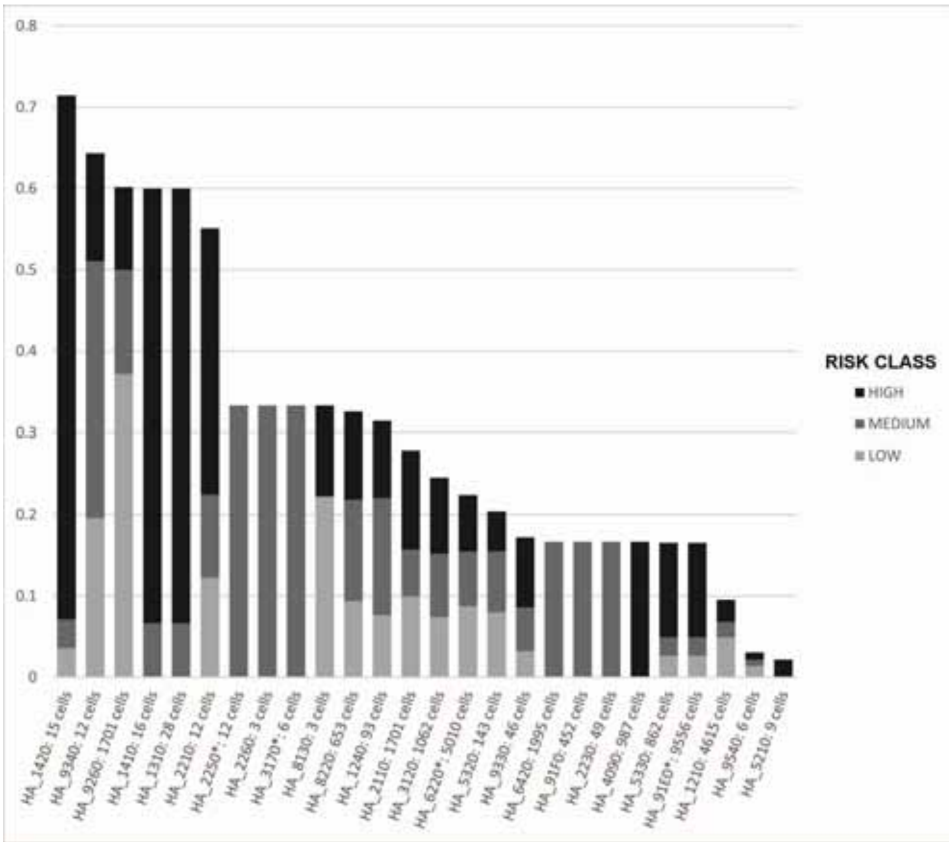

Figure S4 Level of risk for habitat’s type, across all 26 terrestrial habitats. Habitat codes according 92/43 Directive, particularly first digit refers to: 1 and 2 = coastal habitats, 3 = aquatic habitats, 4 and 5 = shrublands; 6 = grasslands, 8 = rocky habitats, 9 = forest habitats. For further details, see Appendix 2.

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6. CONCLUSIONS

The researches collected in this thesis allowed for a better understanding of the process of Plant Invasion in act in the Tuscan Archipelago. We faced this phenomenon adopting a multifocal approach, aiming to embrace the problem from different points of view, but depicting a coherent course that starting from a general improvement of the basic knowledge, passing through a better understanding of the ecological processes in act, would bring to the development of useful tools of management.

The investigations aimed to the characterization the alien flora of the Tuscan Archipelago allowed us to produce an updated checklist, representing a nodal point for the management of the alien plant in TANP. Certainly this is not an arrival point, indeed occurrences and introduction of IAS in the ecosystems will increase in the future, and it appear quite probably that in the time I am writing this thesis new species are introduced in TANP, and in future new update will be needed to maintain keep abreast our knowledge on the alien flora of TANP. However, this contribution represents an essential starting point, allowing to record the current situation, and to build any management strategy. One of the main findings that we highlighted subsequently to the production of the checklist is the increases of magnitude of intentional introduction of potentially invasive plants in TANP, mainly of ornamental plants. This trend, as well as the general increase in species introduction in last 30 years, is a foreseeable consequence of the deep socio-economic changes that involved the Archipelago and more and more islands ecosystems in the last half century. The shift from a historical land-use based on agriculture, livestock and forest exploitation to a model based on tourism, appears as one of the major driving forces in shaping insular ecosystem (Papayannis and Soroeou 2008). Tourism and exploitation of natural assets (for example throughout ecotourism), obviously represent extremely valuable resources for local economies, including fundraising for conservation projects, nevertheless they also raise several issues related to their sustainability (Brundu 2013 and references therein). Strong enforcement of laws and regulations and an intensive educational effort are needed to control IAS introductions in future and to cope with the problems related to this species, especially in this changing context.

The impacts of invasive species on islands have been greatly documented up to now. Our results on both the species we analyzed (*Acacia dealbata* and *A. pycnantha*) highlight the presence of strong ecological impacts on native ecosystems. The displacement of native vegetation led in both case to a strong drop in local biodiversity. The case of *A. dealbata* showed how deep can be the changes in the ecosystem exerted by such “transformer” species. This species affected strongly plants communities and modify deeply chemical and biotical assets of soils. According Hulme et al. (2013) a particular care should be given aiming to relate also species functionality in the evaluating and predicting the changes that they can exert on invaded ecosystems. This is a key case, where a N-fixing species was able to modify and transform the environment. Also the case of *A. pycnatha* allowed to stress these tough trends. The loss of species and the resulting tendency to an impoverishment of understory plant communities in the area invaded by this species is again an evidence of how transformer species can modify the ecosystem. Regarding the last species, further research are needed to better understand the drivers of the process of species loss, investigating the presence of potential allelopathies, the effect of the plant on soil chemistry and on soil biota.

The example of these species applies for many other species. Plants such as *Opuntia spp*, *Carpobrotus spp.* and *Eucalyptus spp.*, have been largely introduced in past for forestry and ornamental purposes, and still the lack of forestry or introduction policies, strongly undermine the conservation of insular ecosystems. Many of these species are still planted mainly in gardens and landscaping, but sometimes even with the purpose of promoting the recovery of native vegetation, also in extremely fragile context (i.e *Carpobrotus spp* used to stabilise sand dunes) (Brundu et al., 2013). Here again raise the need of adequate policies and conservation strategies facing alien plants introduction and exploitation for several uses.

All these considerations finally drove us to the third part of our work, representing an attempt to face risk management of plant invasion in these pleasing islands. Throughout the adoption of Risk Assessment for invasive species we prioritized the alien flora of the Tuscan Archipelago. Particularly the EPPO Prioritizations Process was chosen to supply a prioritization of alien plants to the Tuscan Archipelago National Park managers. We highlighted around 50 plants that should be considered a priority in intervention and monitoring actions by TANP. Moreover, thanks to the scoring system produced by EPP we ranked the species, allowing to concentrate the control efforts on more potentially harmful species, such as *Ailanthus altissima*, *Cortaderia selloana*, *Oxalis pes-caprae*, *Acacia dealbata*, *Melia azedarach*, *Carpobrothus acinaciformis*, *C. edulis* and several *Opuntia species*. These results underlined the needs for eradication and control actions for certain species, to flank the actions already undertaken for some species across the islands of TANP (i.e.

Carpobrothus spp., *Ailanthus altissima* – within the UE LIFE projects LIFE04NAT/IT/000172 and LIFE08 NAT/IT/000353*)*. The latest UE LIFE project funded in TANP (LIFE13-NAT_IT_000471), also thanks to the information collected within this thesis, focuses on further eradication of species such as *Opuntia spp.* and *Agave spp*. And *Oxalis pes-caprae* from Montecristo, and on control of *Eucalyptus camaldulensis* in Pianosa as well as the eradication of *Carpobrotus spp.* from the island of Giannutri. Beyond these eradication projects, a particular attention should be paid in policies and strategy, for example with the encouragement of best practices for forestry and gardening and educational efforts, aiming to prevent further utilization of harmful species (Niemiera and Von Holle 2009). The assessment of risk of invasion on the habitats highlighted some focal situation in the Island of Elba, and represents a good example of how Species Distribution Models can be applied to the framework of biological invasion to produce practical instruments for land managers. The methodology we adopted allowed the identification of those areas exposed to a high risk of invasion, highlighting the contexts that should be selected for a continuous monitoring aimed to a rapid response in case of invasion. Especially in case areas exposed to a high risk host rare habits worthy of conservation (i.e. the case of the complex of habitats of sand dunes of Lacona, Island of Elba), monitoring actions are indeed essential to prevent invasive species to affect valuable biotas.

Concluding, this collection of researches, represent a starting point of a longer process of research and study focused on the understanding and control of plant invasion in such peculiar ecosystems like Mediterranean Island ecosystems, represented by the island of the Tuscan Archipelago. A constant monitoring and updating of the checklist of the alien flora is essential to control the state of the phenomenon, allowing to rapidly record new introduction (and eventually remove new species), and represent the basis for each other action. The study of the impacts and of the ecological processes exerted by alien species has a double scope. On the one hand it is necessary to produce essential data for the Risk Assessment and the evaluation of the potential impacts of alien species. On the other hand these processes represent an exceptional opportunity to understand complex ecological phenomenon, including competition, environmental adaptation and transformation and many others. Finally risk management and risk assessment options are a continuous challenge to face this phenomenon, and scientific communities has the duty to interface with management providing tools and information to counteract the detrimental effects of biological invasions on biodiversity as well as on socio-economic assets, helping to conserve our ecosystem.

Refernces

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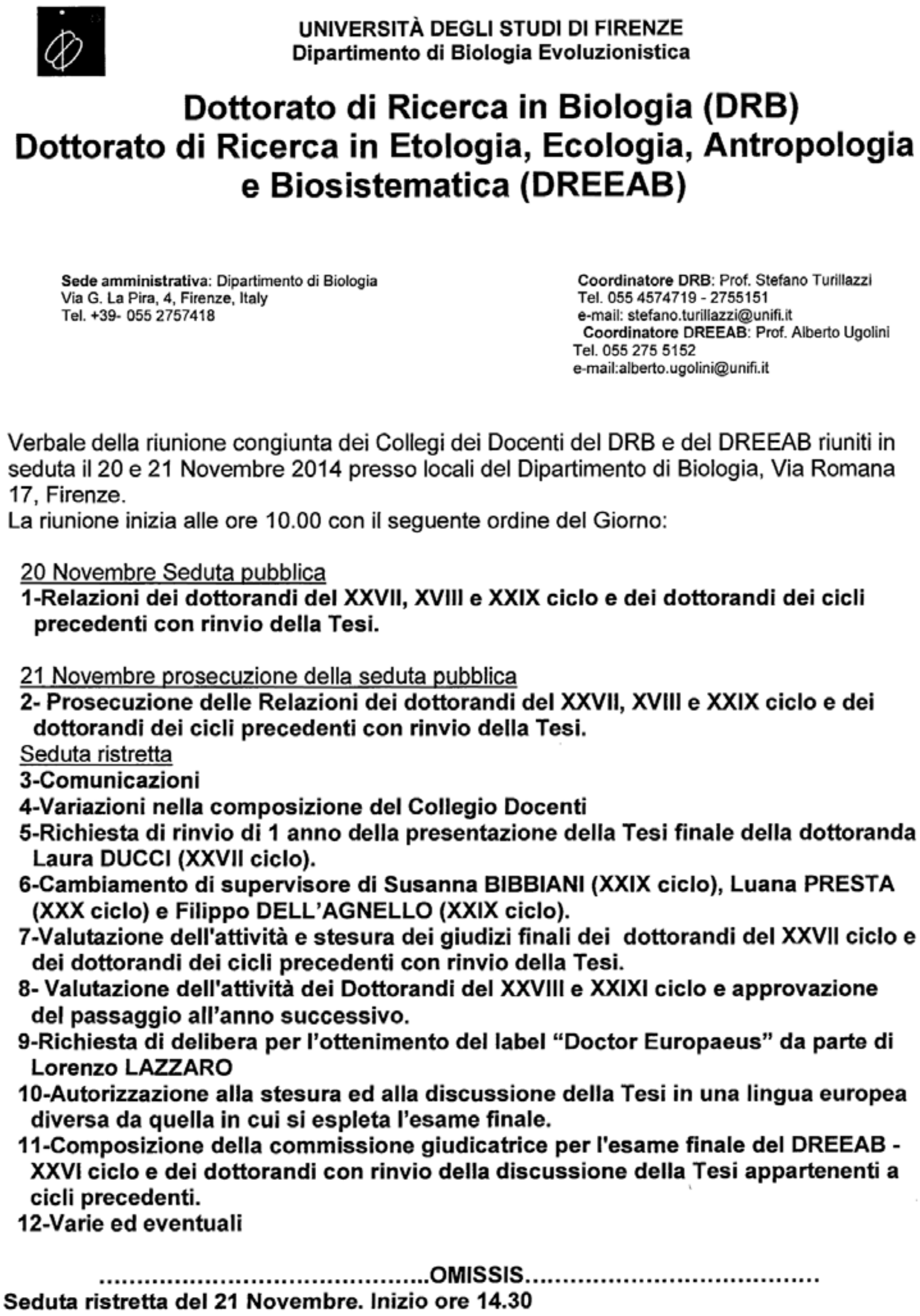
APPENDIX 1 – Presentazione del Collegio dei docenti



UNIVERSITÀ DEGLI STUDI DI FIRENZE
Dipartimento di Biologia Evoluzionistica

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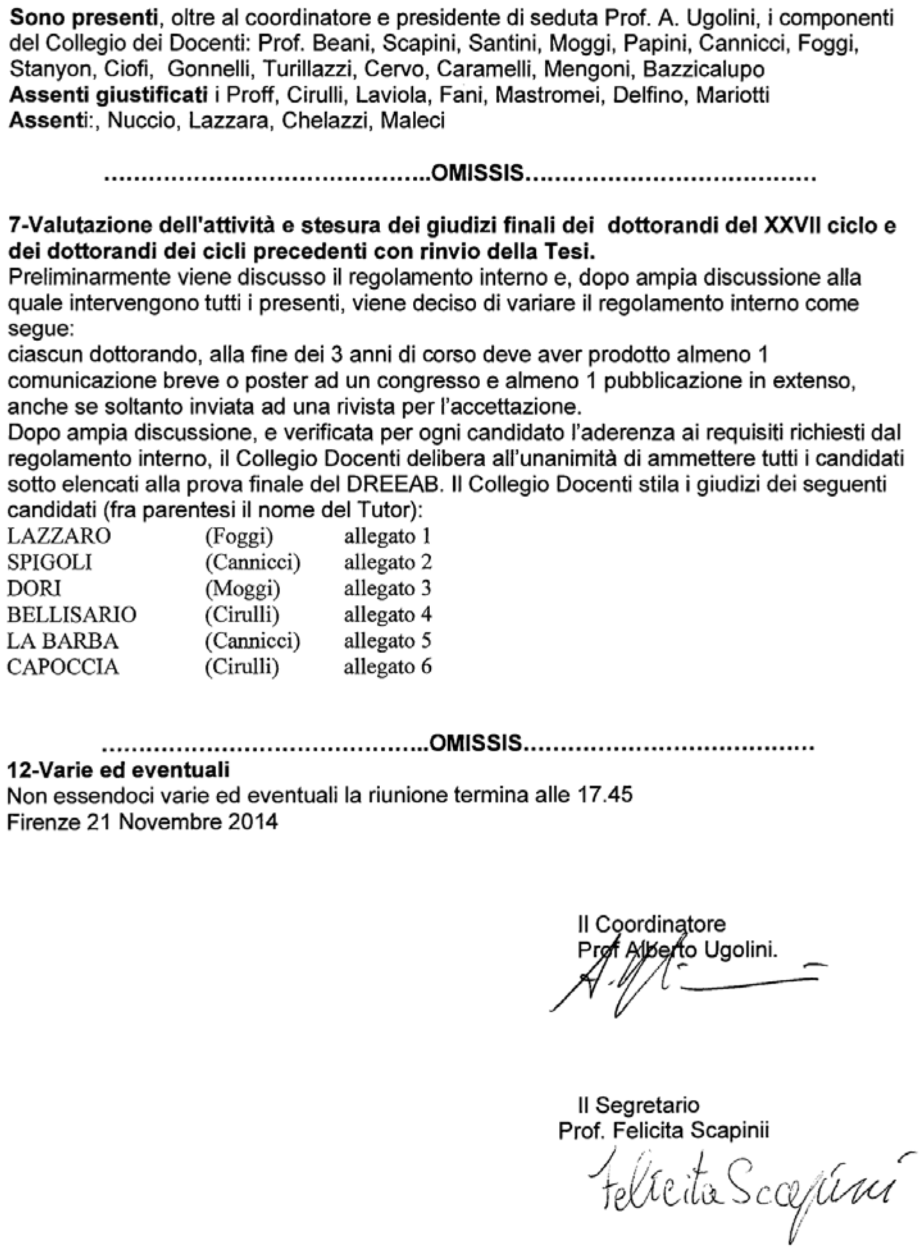
Verbale della riunione congiunta dei Collegi dei Docenti del DRB e del DREEAB riuniti in seduta il 20 e 21 Novembre 2014 presso locali del Dipartimento di Biologia, Via Romana 17, Firenze.
La riunione inizia alle ore 10.00 con il seguente ordine del Giorno:

20 Novembre Seduta pubblica
1-Relazioni dei dottorandi del XXVII, XVIII e XXIX ciclo e dei dottorandi dei cicli precedenti con rinvio della Tesi.

21 Novembre prosecuzione della seduta pubblica
2- Prosecuzione delle Relazioni dei dottorandi del XXVII, XVIII e XXIX ciclo e dei dottorandi dei cicli precedenti con rinvio della Tesi.
Seduta ristretta
3-Comunicazioni
4-Variazioni nella composizione del Collegio Docenti
5-Richiesta di rinvio di 1 anno della presentazione della Tesi finale della dottoranda Laura DUCCI (XXVII ciclo).
6-Cambiamento di supervisore di Susanna BIBBIANI (XXIX ciclo), Luana PRESTA (XXX ciclo) e Filippo DELL'AGNELLO (XXIX ciclo).
7-Valutazione dell'attività e stesura dei giudizi finali dei dottorandi del XXVII ciclo e dei dottorandi dei cicli precedenti con rinvio della Tesi.
8- Valutazione dell'attività dei Dottorandi del XXVIII e XXIXI ciclo e approvazione del passaggio all'anno successivo.
9-Richiesta di delibera per l'ottenimento del label "Doctor Europaeus" da parte di Lorenzo LAZZARO
10-Autorizzazione alla stesura ed alla discussione della Tesi in una lingua europea diversa da quella in cui si espleta l'esame finale.
11-Composizione della commissione giudicatrice per l'esame finale del DREEAB - XXVI ciclo e dei dottorandi con rinvio della discussione della Tesi appartenenti a cicli precedenti.
12-Varie ed eventuali

…………………………………….OMISSIS…………………………………

Seduta ristretta del 21 Novembre. Inizio ore 14.30



Sono presenti, oltre al coordinatore e presidente di seduta Prof. A. Ugolini, i componenti del Collegio dei Docenti: Prof. Beani, Scapini, Santini, Moggi, Papini, Cannicci, Foggi, Stanyon, Ciofi, Gonnelli, Turillazzi, Cervo, Caramelli, Mengoni, Bazzicalupo
Assenti giustificati i Proff, Cirulli, Laviola, Fani, Mastromei, Delfino, Mariotti
Assenti:, Nuccio, Lazzara, Chelazzi, Maleci

……………………………………..OMISSIS…………………………………

7-Valutazione dell'attività e stesura dei giudizi finali dei dottorandi del XXVII ciclo e dei dottorandi dei cicli precedenti con rinvio della Tesi.
Preliminarmente viene discusso il regolamento interno e, dopo ampia discussione alla quale intervengono tutti i presenti, viene deciso di variare il regolamento interno come segue:
ciascun dottorando, alla fine dei 3 anni di corso deve aver prodotto almeno 1 comunicazione breve o poster ad un congresso e almeno 1 pubblicazione in extenso, anche se soltanto inviata ad una rivista per l'accettazione.
Dopo ampia discussione, e verificata per ogni candidato l'aderenza ai requisiti richiesti dal regolamento interno, il Collegio Docenti delibera all'unanimità di ammettere tutti i candidati sotto elencati alla prova finale del DREEAB. Il Collegio Docenti stila i giudizi dei seguenti candidati (fra parentesi il nome del Tutor):

LAZZARO	(Foggi)	allegato 1
SPIGOLI	(Cannicci)	allegato 2
DORI	(Moggi)	allegato 3
BELLISARIO	(Cirulli)	allegato 4
LA BARBA	(Cannicci)	allegato 5
CAPOCCIA	(Cirulli)	allegato 6

……………………………………..OMISSIS…………………………………

12-Varie ed eventuali
Non essendoci varie ed eventuali la riunione termina alle 17.45
Firenze 21 Novembre 2014

Il Coordinatore
Prof Alberto Ugolini.

Il Segretario
Prof. Felicita Scapinii
Felicita Scapini

ALLEGATO 1
Dottorato in Etologia, Ecologia, Antropologia e Biosistematica, XXVII ciclo.
Candidato: Lorenzo Lazzaro. **Tutor**: Bruno Foggi, **Co-Tutor**: Renato Benesperi, Giuseppe Brundu

Lorenzo Lazzaro si è laureato in Scienze Naturali presso L'Università degli Studi di Firenze nel 2007 con voto di 110/110. In seguito ha conseguito la laurea magistrale in Conservazione e Gestione della Natura; Curriculum di Risorse Vegetali nell'aprile 2010 con voto 110/110 e lode e nel 2011 è stato ammesso al corso di Dottorato di ricerca in Etologia, Ecologia, Antropologia e Biosistematica (XXVII ciclo), indirizzo in Biologia Vegetale. Il lavoro di ricerca nell'ambito del dottorato si è incentrato sul problema delle specie vegetali alloctone in Arcipelago Toscano. Le tematiche delle ricerche hanno incluso aspetti distributivi, ecologici incluso i problemi gestionali del fenomeno. In particolare nel primo e nel secondo anno le attività si sono incentrare sulla formazione di un ampio background di letteratura sull'argomento e sulla raccolta di dati, mentre il terzo anno è stato dedicato alla sintesi e pubblicazione dei risultati delle ricerche. Distribuzioni e naturalizzazione delle specie aliene dell'Arcipelago sono state indagate nei primi due anni sia su base bibliografica, che attraverso una importante fase esplorativa sul campo (1, 2). Il lavoro ha portato alla redazione di una checklist della flora aliena dell'Arcipelago Toscano (3). Gli aspetti ecologici hanno riguardato l'indagine degli eventuali impatti dovuti a specie invasive all'Isola d'Elba. Tali studi hanno permesso di rilevare impatti sul chimismo dei suoli e sulle comunità microbiche del suolo e vegetali per *Acacia dealbata* (4) e sugli impatti sulle comunità vegetali su *Acacia pycnantha* (5). Infine per gli aspetti gestionali si è lavorato su la valutazione del rischio di invasione attraverso due approcci mirati alla produzione di due lavori. L'invasività delle specie aliene in Arcipelago Toscano è stata valutata mediante l'uso di procedure di Risk Assessment (6), e il rischio di invasione da parte delle specie aliene sugli habitat meritevoli di conservazione mediante l'utilizzo di Modelli di Habitat potenziale (7). Le tematiche affrontate hanno permesso di mettere in luce diversi aspetti della biologia delle invasioni in ambiente insulare mediterraneo, fornendo sia apporti teorici alle tematiche sugli impatti delle specie invasive che apporti tecnici utili ai fini della conservazione, quali il monitoraggio delle distribuzioni attuali e aspetti di valutazione del rischio e prioritizzazione degli sforzi di gestione. Oltre alle attività strettamente inerenti le principali attività ricerca il candidato ha partecipato ad altri lavori, sempre legati alla tematica delle specie invasive, legati a test di germinabilità su semi di *Robinia pseudoacacia* (8) e all'individuazione di pattern di Biotic Homogenization nella flora lichenica di aree invase sempre dalla robinia (9). La formazione del candidato, oltre alle attività previste dal dottorato, ha incluso la partecipazione a corsi di approfondimento e di studio all'estero, quali la partecipazione ad un training school di 4 giorni a Parigi (Francia) sulle tematiche del Risk Assessment e un corso di Ecologia Quantitativa di 4 mesi presso l'università di Ceske Budejopvice (Repubblica Ceca). In particolare quest'ultima attività ha avuto a corollario, oltre che lo studio approfondito di metodi di analisi statistica successivamente applicati in gran parte dei lavori, anche la stesura di un lavoro su tematiche ecologiche di competitività e facilitazione nelle comunità vegetali (10).
Le attività del dottorando nel corso del triennio hanno portato alla partecipazione a convegni con poster, a seminari nel corso di laurea magistrale in Scienze dalla natura e dell'uomo e bel master STRIVE organizzato dal dipartimento di Biologia, nonché alla produzione finale di alcuni articoli.

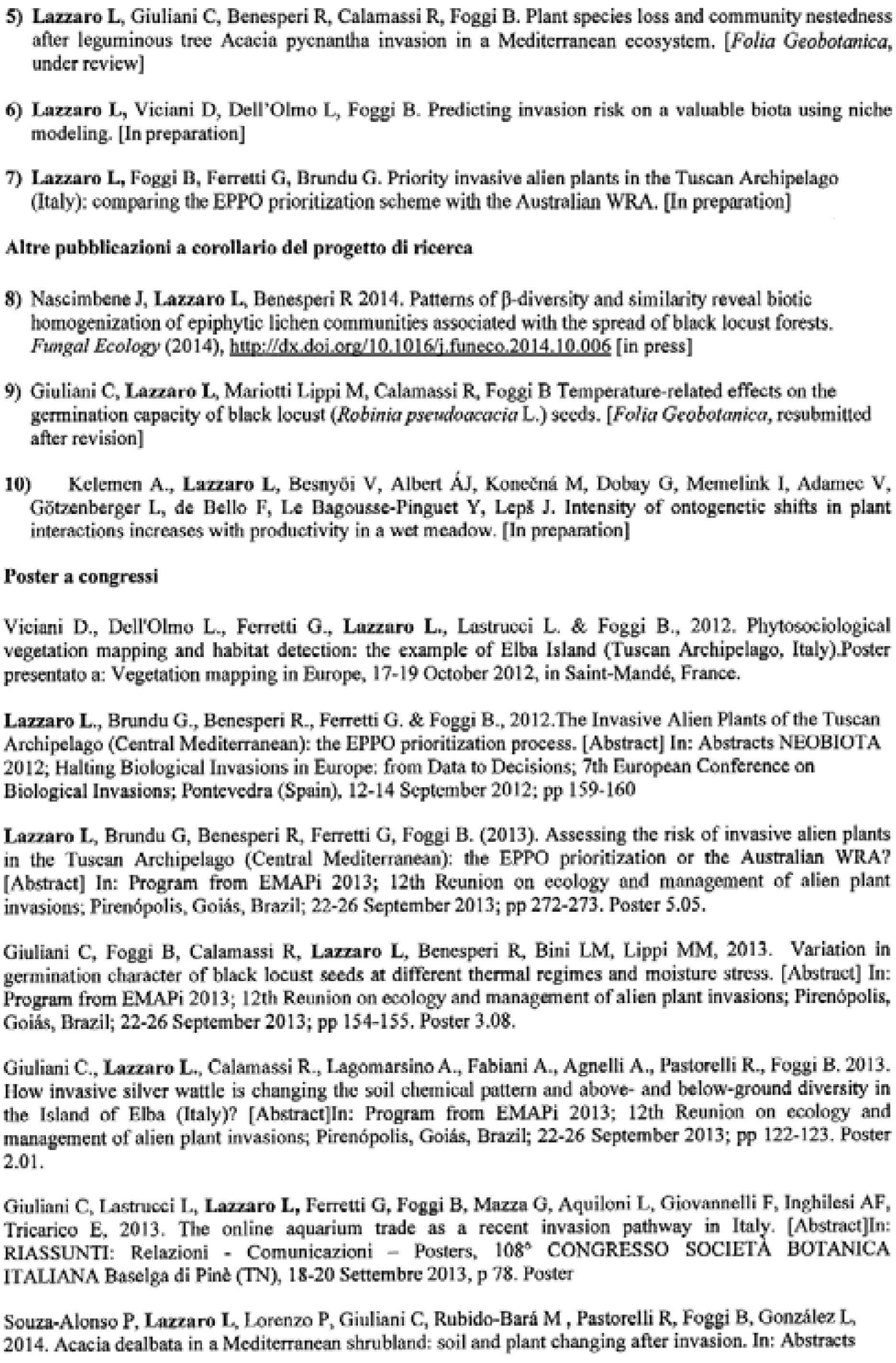
Pubblicazioni inerenti il progetto di ricerca tema del dottorato

1) **Lazzaro L**, Ferretti G, Galasso G, Lastrucci L, Foggi B 2013. Contributo alla conoscenza della flora esotica dell'Arcipelago Toscano, Italia. *Atti Società Italiana di Scienze Naturali Museo Civico di Storia Naturale di Milano*. 154(1):3–24.

2) Ferretti G, **Lazzaro L**, Giuliani C, Foggi B 2013. Secondo contributo alla conoscenza della flora esotica dell'Arcipelago Toscano, Italia. *Atti Società Italiana di Scienze Naturali Museo Civico di Storia Naturale di Milano* 154(2): 115–130.

3) **Lazzaro L**, Ferretti G, Giuliani C, Foggi B 2014. A checklist of the alien flora of the Tuscan Archipelago (Italy). *Webbia* 69: 157–176.

4) **Lazzaro L**, Giuliani C, Fabiani A, Agnelli AE, Pastorelli R, Lagomarsino A, Benesperi R, Calamassi R, Foggi B 2014. Soil and plant changing after invasion: The case of Acacia dealbata in a Mediterranean ecosystem. *Science of The Total Environment* 497-498: 491–498.



5) **Lazzaro L**, Giuliani C, Benesperi R, Calamassi R, Foggi B. Plant species loss and community nestedness after leguminous tree Acacia pycnantha invasion in a Mediterranean ecosystem. [*Folia Geobotanica*, under review]

6) **Lazzaro L,** Viciani D, Dell'Olmo L, Foggi B. Predicting invasion risk on a valuable biota using niche modeling. [In preparation]

7) **Lazzaro L,** Foggi B, Ferretti G, Brundu G. Priority invasive alien plants in the Tuscan Archipelago (Italy): comparing the EPPO prioritization scheme with the Australian WRA. [In preparation]

Altre pubblicazioni a corollario del progetto di ricerca

8) Nascimbene J, **Lazzaro L**, Benesperi R 2014. Patterns of β-diversity and similarity reveal biotic homogenization of epiphytic lichen communities associated with the spread of black locust forests. *Fungal Ecology* (2014), http://dx.doi.org/10.1016/j.funeco.2014.10.006 [in press]

9) Giuliani C, **Lazzaro L**, Mariotti Lippi M, Calamassi R, Foggi B Temperature-related effects on the germination capacity of black locust (*Robinia pseudoacacia* L.) seeds. [*Folia Geobotanica*, resubmitted after revision]

10) Kelemen A., **Lazzaro L**, Besnyői V, Albert ÁJ, Konečná M, Dobay G, Memelink I, Adamec V, Götzenberger L, de Bello F, Le Bagousse-Pinguet Y, Lepš J. Intensity of ontogenetic shifts in plant interactions increases with productivity in a wet meadow. [In preparation]

Poster a congressi

Viciani D., Dell'Olmo L., Ferretti G., **Lazzaro L.**, Lastrucci L. & Foggi B., 2012. Phytosociological vegetation mapping and habitat detection: the example of Elba Island (Tuscan Archipelago, Italy).Poster presentato a: Vegetation mapping in Europe, 17-19 October 2012, in Saint-Mandé, France.

Lazzaro L., Brundu G., Benesperi R., Ferretti G. & Foggi B., 2012.The Invasive Alien Plants of the Tuscan Archipelago (Central Mediterranean): the EPPO prioritization process. [Abstract] In: Abstracts NEOBIOTA 2012; Halting Biological Invasions in Europe: from Data to Decisions; 7th European Conference on Biological Invasions; Pontevedra (Spain), 12-14 September 2012; pp 159-160

Lazzaro L, Brundu G, Benesperi R, Ferretti G, Foggi B. (2013). Assessing the risk of invasive alien plants in the Tuscan Archipelago (Central Mediterranean): the EPPO prioritization or the Australian WRA? [Abstract] In: Program from EMAPi 2013; 12th Reunion on ecology and management of alien plant invasions; Pirenópolis, Goiás, Brazil; 22-26 September 2013; pp 272-273. Poster 5.05.

Giuliani C, Foggi B, Calamassi R, **Lazzaro L**, Benesperi R, Bini LM, Lippi MM, 2013. Variation in germination character of black locust seeds at different thermal regimes and moisture stress. [Abstract] In: Program from EMAPi 2013; 12th Reunion on ecology and management of alien plant invasions; Pirenópolis, Goiás, Brazil; 22-26 September 2013; pp 154-155. Poster 3.08.

Giuliani C., **Lazzaro L.**, Calamassi R., Lagomarsino A., Fabiani A., Agnelli A., Pastorelli R., Foggi B. 2013. How invasive silver wattle is changing the soil chemical pattern and above- and below-ground diversity in the Island of Elba (Italy)? [Abstract]In: Program from EMAPi 2013; 12th Reunion on ecology and management of alien plant invasions; Pirenópolis, Goiás, Brazil; 22-26 September 2013; pp 122-123. Poster 2.01.

Giuliani C, Lastrucci L, **Lazzaro L,** Ferretti G, Foggi B, Mazza G, Aquiloni L, Giovannelli F, Inghilesi AF, Tricarico E, 2013. The online aquarium trade as a recent invasion pathway in Italy. [Abstract]In: RIASSUNTI: Relazioni - Comunicazioni – Posters, 108° CONGRESSO SOCIETÀ BOTANICA ITALIANA Baselga di Pinè (TN), 18-20 Settembre 2013, p 78. Poster

Souza-Alonso P, **Lazzaro L**, Lorenzo P, Giuliani C, Rubido-Bará M , Pastorelli R, Foggi B, González L, 2014. Acacia dealbata in a Mediterranean shrubland: soil and plant changing after invasion. In: Abstracts

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Comunicazioni orali a convegni e seminari

Lazzaro L, Ferretti G, Foggi B. Monitoraggio flora e vegetazione Monitoring of flora and vegetation. WORKSHOP Le specie aliene invasive, una grave minaccia per i sistemi insulari: esperienze di gestione a confronto *Invasive alien species, a major threat to insular ecosystems: comparing management options and experiences* Portoferraio, 31 marzo/*march* – 2 aprile/*april* 2014

Lazzaro L. Invasività e Invasibilità nella gestione delle piante aliene invasive - Weed Risk Assessment, Modelli Predittivi Ecologici, Impatti delle IAS: esperienze in toscana. Seminario presentato al corso di laurea di Scienze della Natura e dell'Uomo nel corso di Conservazione delle risorse vegetali negli anni 2013, 2014; al master STRIVE del dipartimento di Biologia dell'università di Firenze e al master di Gestione della Fauna dell'Università Di Parma.

Comunicazione scientifica e relazioni tecniche

Camilla Gotti, Mario Cozzo, Adriano De Faveri, Marco Zenatello, Nicola Baccetti, **Lorenzo Lazzaro**, Giulio Ferretti, Bruno Foggi Il monitoraggio della fauna e della flora a Montecristo. All'interno di: I Quaderni del Parco, documenti tecnici volume 2. Report Progetto Life+ Montecristo 2010, pp 54-69

Foggi B, Calamassi R, **Lazzaro L.** Monitoraggio degli interventi su pino d'Aleppo e piante esotiche invasive a Pianosa All'interno di: I Quaderni del Parco, documenti tecnici volume 2. Report Progetto Life+ Montecristo 2010, pp 70-74

B. Foggi, **L. Lazzaro**, G. Ferretti, C. Giuliani, L. Dell'olmo. Specie esotiche vegetali dell'arcipelago toscano check list, status e hot spot di vulnerabilita'. Progetto Co.R.E.M. CUP E79E1000012000 – CIG 3144073b1b. Relazione tecnica.