CrossMark

REVIEW PAPER

Here to stay. Recent advances and perspectives about *Acacia* invasion in Mediterranean areas

Pablo Souza-Alonso[1] · Jonatan Rodríguez[1] · Luís González[1] · Paula Lorenzo[2]

Abstract

• *Key message* The above- and belowground impacts due to *Acacia* invasions have been described in detail over the last 25 years. Future research should focus on the early detection and prevention of new *Acacia* introductions and on a cost-effective and sustainable management of the novel ecosystems resulting from invasions.

• *Context* Invasive alien plants (IAPs) strongly alter ecosystems reducing biodiversity, modifying ecosystem services and increasing negative impacts at social and economic level. Among invasive taxa, *Acacia* is a highly problematic genus worldwide. In fact, almost 500 papers have been published on several aspects of *Acacia* invasions for the last 25 years.

• *Aims* We aim at reviewing the current knowledge on the consequences of the invasion by Acacia genus in Mediterranean ecosystems. We also collect and propose different approaches for the management and recovery of invaded areas and suggest future perspectives on Acacia research.

• *Methods* We compile, summarise and discuss recent findings on physicochemical, ecological, microbiological and socioeconomic aspects of invasion related to Australian acacias (Acacia dealbata, Acacia longifolia, Acacia mearnsii, Acacia saligna and Acacia melanoxylon) focusing on Mediterranean areas.

• *Results* Acacia invasion generally entails soil physicochemical alterations and changes in microbial function and structure. Consequences such as the decreased biodiversity, altered ecosystem structure, larger seed banks dominated by invasive species, new biotrophic relationships or alterations in water availability and fire regimes suggest that acacias are locally creating novel ecosystems.

• *Conclusions* Forecasting invasions, modelling and managing ecosystems dominated by acacias are challenging tasks that should be addressed in the future, since climatic conditions and intensification in land uses are increasing the likelihood of *Acacia* invasions in Mediterranean areas. Unsuccessful management actions suggest that restoration should be meticulously monitored, but the magnitude of invasion or the inconsistency of economic investment indicate that eradication is often unfeasible. Alternatively, novel integrative and cost-effective solutions including the collaboration of society, politicians and stakeholders are necessary to prevent new introductions and achieve sustainable control of acacias. There is a growing interest in applied research on the valorisation or novel uses for acacias and their residues that result in economic benefits.

Handling Editor: Andreas Bolte

Contribution of the co-authors Pablo Souza-Alonso: conceived the paper, obtained information, coordinate the review, wrote the first draft and edited the manuscript.
Paula Lorenzo: obtained information, wrote different sections and contributed to correct and organise the manuscript.
Jonatan Rodríguez: obtained information and contributed to correct different sections.
Luís González: contributed to correct and organise the manuscript.

✉ Pablo Souza-Alonso
souzavigo@gmail.com

Jonatan Rodríguez
jonatan@uvigo.es

Luís González
luis@uvigo.es

Paula Lorenzo
paulalorenzo@uc.pt

[1] Plant Biology and Soil Science Department, University of Vigo, 36310 Vigo, Spain

[2] Centro de Ecologia Funcional - CEF, Departamento de Ciências da Vida, Faculdade de Ciências e Tecnologia, Universidade de Coimbra, 3000-456 Coimbra, Portugal

INRA
SCIENCE & IMPACT

🖉 Springer

1 Introduction

Humans induce rapid changes to the environment, including the alteration of major biogeochemical cycles, land surface transformation, changed atmospheric composition and evolution patterns. Such changes are currently taking place at unprecedented rates in the period recently defined as the Anthropocene (Steffen et al. 2007; Lewis and Maslin 2015). Numerous human activities act as driving forces of environmental change by removing physical, biotic and geographical barriers that facilitate plant species movement, which is a main risk factor for biodiversity. Invasive alien plants (IAPs), defined as plant species producing large reproductive progeny that spread over considerable distances from parent plants (Richardson et al. 2000a), are currently a priority research objective of the European Commission (EC Regulation 1143/2014).

Along the wide range of genera containing species classified as invaders, *Acacia* is one of the most controversial and studied genus across the world (Murphy 2008). Currently, 24 *Acacia* species are confirmed as invasive worldwide (Richardson and Rejmánek 2011; Lorenzo and Rodríguez-

Echeverría 2015). *Acacia sensu lato* is a cosmopolitan genus with polyphyletic origin, but in this paper, we exclusively focus on the Australian Acacia subgenus *Phyllodineae* - *Acacia s.s.*- (Kyalangalilwa et al. 2013) due to their invasiveness. The movement of acacias to other continents began in the late 1700s (Carruthers et al. 2011), but unprecedented dispersal rates have occurred in the latest two centuries. Albeit Australian acacias are now occurring worldwide, they are more frequently invasive in Mediterranean areas (Fig. 1). Studies on *Acacia* invasions were compiled in a special issue covering not only biological and ecological but also social, economic and ethical perspectives (human-mediated introductions of Australian acacias—a global experiment in biogeography, Diversity and Distributions 2011). At the same time, Lorenzo et al. (2010a) published a review mainly focused on hypotheses explaining the invasive success of *Acacia dealbata* in Europe. These authors suggest that *A. dealbata* not only takes advantage from environmental disturbances, but also possesses high clonal growth and allelopathic ability that reduce native biodiversity in the understory. However, due to the increasing number of studies recently conducted in Mediterranean areas (see also Fig. 1), we consider that an update of the current knowledge on the consequences of *Acacia* invasion at these regions is required. Here, our objective is to not only summarise recent findings (including biological, ecological, physicochemical, microbiological or socioeconomic aspects of invasion) but also to complement
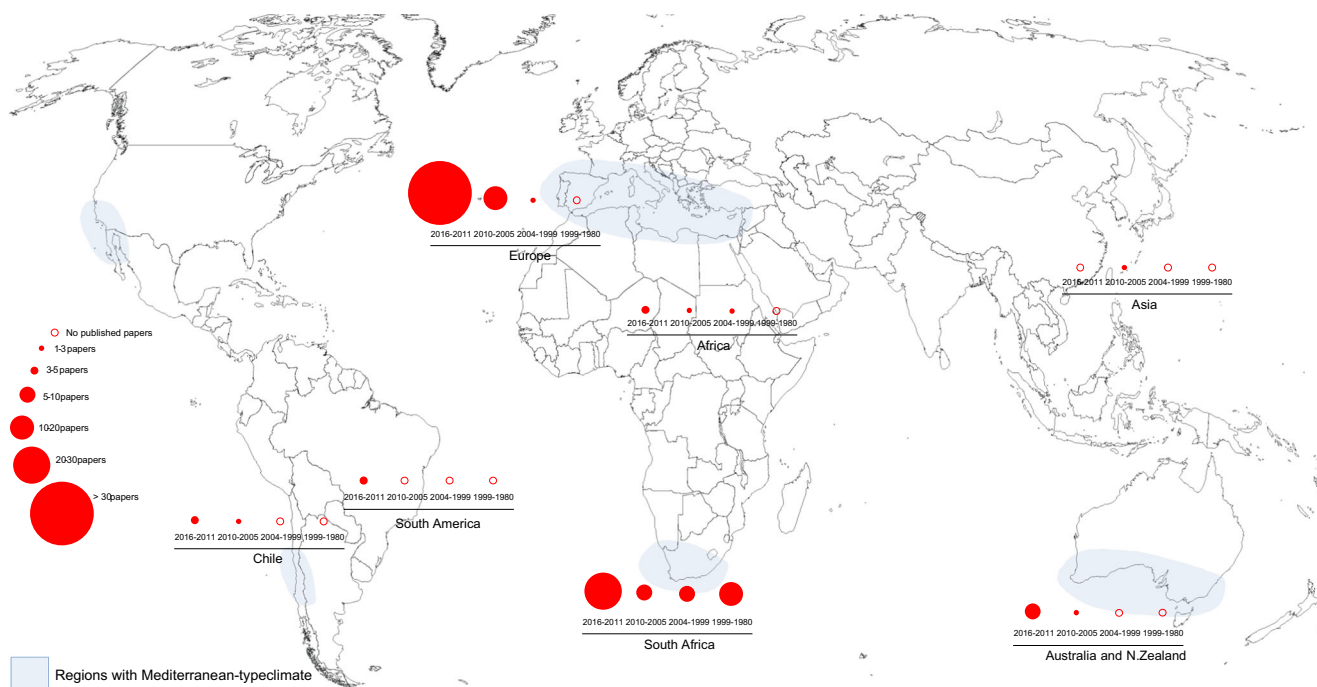
Fig. 1 Scheme representing the number of papers including highly invasive acacias considered: *Acacia dealbata, A. longifolia, A. mearnsii, A. saligna* and *A. melanoxylon*, through different areas across the world. Map is created based on information (*n* = 214 manuscripts) after the search in SCOPUS by using the key terms *Acacia* and *plant* and *invas** (TITLE-ABS-KEY)+ and "*Acacia dealbata*" or "*Acacia longifolia*" or "*Acacia saligna*" or "*Acacia melanoxylon*" or "*Acacia mearnsii*". Systematical reviews on the *Acacia* invasive process were not included unless they were strictly focused on a specific area. Note the different scale for the last interval (1999–1980)

and extend previous information related to the most problematic acacias in Mediterranean areas. We also discuss future perspectives on research, management and recovery of invaded areas.

1.1 Major problematic acacia species, introduction and current distribution

Although there is an important number of invasive species within *Acacia* genus, we focused on *Acacia dealbata* Link, *Acacia longifolia* (Andrews) Willd., *Acacia mearnsii* De Wild., *Acacia saligna* (Labill) H. L. Wendl. and *Acacia melanoxylon* R. Br. due to their impacts worldwide (Richardson and Rejmánek, 2011; Lorenzo and Rodríguez-Echeverría 2015). *Acacia cyclops*, also considered as an IAP, was not included since available data is almost exclusively based on information from the Cape Region, South Africa (Higgins et al. 1999, 2001).

Specific characteristics of these acacias, such as adaptability to many environmental conditions, easy germination and growth, good survival and rapid growth rates, ornamental value or wood quality, have determined their current distribution (Maslin and McDonald, 2004). In Europe and other Mediterranean areas, some uses of acacias as wood and timber production (Griffin et al. 2011), the perfume industry (Perriot et al. 2010; Kull et al. 2011), stabilisation of dunes and avoidance of sand erosion (Marchante et al. 2003; Cohen et al. 2008; Del Vecchio et al. 2013) or to stabilise slopes derived from the railway construction (Kull et al. 2007), played a significant role in their introduction. Consequently, a wide range of Mediterranean biomes are currently threatened by acacias, such as riparian habitats, shrublands, fynbos, sclerophyllous forests, mixed forests, grasslands and prairies, coastal areas and sand dunes, riverlands and watercourses, islands, agricultural fields or tree plantations (Le Maitre et al. 2000; Marchante et al. 2003; Rodríguez-Echeverría et al. 2009; Lorenzo et al. 2010a, b; Crous et al. 2012; Boudiaf et al. 2013; Hernández et al. 2014; Lazzaro et al. 2014; Celesti-Grapow et al. 2016). In fact, invasive acacias have been also defined as *transformers*, those species that "substantially change the character, condition, form or nature of ecosystems, becoming active agents in region-forming processes" (Richardson et al. 2000a; Marchante et al. 2011a).

1.2 Human perception of invasive acacias

"A fascinating story to be told regarding what transpires when an environmental scientist's problem is a rural community's livelihood" is how Kull et al. (2011) summarises the contradictory perception of exotic acacias when they represent an economic resource and, at the same time, an ecological threat in the introduced ranges. Human perception of invasive acacias is strongly influenced by biophysical, familiarity, social

variables and socioeconomic circumstances (Tassin et al. 2009a; Kull et al. 2011, 2007). In many countries, large parts of the population have positive perceptions about invasive acacias that are largely cultivated with profitable uses such as construction materials, heat source or medicinal compounds for rural communities (de Neergaard et al. 2005; Wintola et al. 2017). For example, *A. dealbata* is highly valued by local communities of Spain, Portugal and France, where festivals have been celebrated in its honour for almost 50 years (Afonso 2012). In France, some villages such as Mandelieu-la-Napoule or Biot, both at the Côte d'Azur, have celebrations of *A. dealbata* that have continued for more than 80 and 60 years, respectively. On the other hand, when economic activities are affected by *Acacia* invasion, such as forestry or citrus cultivation (Kull et al. 2011), these species are being recognised as problematic.

2 New insights into traits that promote invasion

2.1 Genetics, phenotype and physiology

The size of the genomic pool has been suggested as a factor promoting invasion (Grotkopp et al. 2004; Kubešová et al. 2010). However, univariate analyses comparing the genome size of 92 acacias introduced outside their native range—21 invasive, 71 non-invasive—did not detect any difference between the genome size based on their invasive character (Gallagher et al. 2011). In addition, low levels of genetic diversity in the introduced areas compared to native areas are not necessarily related to a reduction in the invasion success and vice versa (Harris et al. 2012).

The amplitude of the native range is generally considered as an important predictor of invasiveness as it reveals the adaptation to a wide range of environmental conditions and leads to a large risk of propagation and dispersal by humans (Goodwin et al. 1999). Life history traits, such as tree height and sprouting ability, have an important weight in invasiveness predictive models (Gallagher et al. 2011; Gibson et al. 2011). However, their importance decreases when human factors are included. In fact, human use is one of the most important predictors of *Acacia* invasiveness (Castro-Díez et al. 2011).

Ecophysiological traits can be as important as morphological traits in explaining invasiveness. Once seeds reach the soil, acacias are provided with mechanisms to outcompete native species. For example, in saline water-stressed environments, *A. longifolia* seeds and seedlings present increased intracellular ion concentrations, efficient nitrogen uptake, defence against superoxide radicals and high tolerance to a wide range of salt concentrations compared to native species (Morais et al. 2012; Morais and Freitas 2012). Godoy et al. (2011) indicated that the photosystem II (PSII) activity of

A. melanoxylon performs better with higher leaf temperature than that of natives under water stress. This fact could reflect a higher thermostability of the PSII or, on the contrary, a better acclimation and thus, efficiency of the entire photosynthetic process in arid or Mediterranean-type ecosystems. Under experimental conditions, an increase in CO_2 has been related to higher growth rates, final weight and increased N-fixation rates of *A. melanoxylon* (Schortemeyer et al. 2002). Consequently, if N supply is also increased, dry biomass, CO_2 assimilation, foliage thickness and density are significantly enhanced (Schortemeyer et al. 1999). In a expected global warming scenario with higher temperatures and CO_2 levels (IPCC 2013), with acacias growing at higher rates and producing canopies with denser foliage, reducing light availability for understory species, the invasiveness of these species could be severely increased. However, the benefit from new climatic conditions is not clear, at least for *A. dealbata*. González-Muñoz et al. (2014) predicted a decline of its growth in the Iberian Peninsula based on climate-growth patterns and climatic models. Conversely, using species distribution models, habitat connectivity and protected areas layers, Vicente et al. (2016) forecasted an increasing land exposition and connectivity between suitable areas for *A. dealbata* due to climate change.

2.2 Reproductive features

Reproduction by sprouting facilitates the establishment of clonal populations (Lorenzo et al. 2010a; Fuentes-Ramírez et al. 2011; Rodríguez et al. 2017). In fact, the proportion of sprouting species is higher among invasive than non-invasive acacias (Gibson et al. 2011). Invasive acacias also reach reproductive maturity earlier (<2 years) than non-invaders (Gibson et al. 2011). *Acacia dealbata* and *A. mearnsii* tend to have higher levels of self-compatibility, suggesting that the ability to self-fertilise may favour its invasiveness (Gibson et al. 2011). Indeed, *A. dealbata* has the capacity to produce progeny by autonomous self-pollination (Rodger and Johnson, 2013). Besides *A. dealbata*, also *A. longifolia* and *A. melanoxylon* showed low level of spontaneous self-pollination allowing them to produce viable offspring in Portugal (Correia et al. 2014). Nevertheless, in the native range of *A. dealbata*, there was little evidence of elevated inbreeding influencing its progeny (Broadhurst et al. 2008). In addition, *A. saligna* has a mixed mating system, preferential outcrossing, but also with a certain level of selfing (George et al. 2008). The ecological function of self-pollination and its role in invasiveness is highly dependent on the trade-off between the benefits of the absence of compatible mates and costs, such as the inbreeding depression (characterised by a reduction in growth and progeny survival). Self-pollination could be a valuable tool to produce offspring under circumstances

that severely constrain plant survival (e.g. isolated areas, absence of pollinators, mate limitation). Interestingly, *A. mearnsii* showed both sexual and asexual reproduction depending on the environmental conditions in the non-native range, showing preference for sprouting in disturbed areas and seed-based reproduction in undisturbed sites (Eilu and Obua 2005).

Acacias and pollinators Despite their ability to self-fertilisation, acacias are pollinated by generalist insects and they usually require the presence of pollination vectors to achieve significant seed production (Correia et al. 2014). In fact, low pollen/ovule ratio supports the compatibility with dependence on animal pollen vectors (Gibson et al. 2011). Reproductive success is often maximised by the synchronised and massive opening of flowers both within a single individual and local populations (Stone et al. 2003), which may interfere with the normal relationship between native species and their pollinators. In South Africa, Gibson et al. (2013) indicated that flower visitation to native plants was reduced due to the presence of *A. saligna*. Nevertheless, despite the massive flowering of *A. dealbata* and *A. longifolia*, native plant species attained similar or even higher visitation rates in Portugal (Montesinos et al. 2016). Complementary low temperatures and high relative humidity during winters in the Northern hemisphere favour polyad viability and pollen tube development (Beck-Pay 2012).

Seed production, dispersal and germination Seed production is suggested as a factor promoting the invasion of acacias (Castro-Díez et al. 2011). In the introduced range, *A. dealbata* and *A. longifolia* escape pre-dispersal predation and display a higher production of fully developed seeds per fruit (*A. longifolia*) or per tree (*A. dealbata*), accompanied with larger size of individual seeds (Correia et al. 2016). However, rare and widespread acacias have similar levels of seed production (quantitatively and qualitatively), indicating that, in some cases, the level of seed development and release does not necessarily determine plant abundance (Buist 2003). Nevertheless, massive seed production and accumulation is highly variable within acacias (Gibson et al. 2011). Once released, seeds can be dispersed by water or wind, but also through myrmecochory (seeds with elaiosomes) or ornithochory (seeds with arils) (French and Major 2001; Richardson and Kluge 2008; Marchante et al. 2010; Montesinos et al. 2012), remaining viable for up to 150–200 years (Daws et al. 2007; Leino and Edqvist 2010).

Fire stimulates seed germination of several invasive acacias such as *A. melanoxylon*, *A. dealbata* and *A. saligna* (García et al. 2007; Lorenzo et al. 2010a; Wilson et al. 2011). Additionally, butenolide, a chemical compound isolated from smoke, may have a significant positive effect on the post-fire seedling ecology of *A. mearnsii* (Kulkarni et al. 2007). The

stimulating effect of fire has important ecological implications since fire may eliminate native seeds from the surface layer, favouring the germination of resistant acacia seeds and thus, the success of the invasion (Richardson and Kluge 2008; Le Maitre et al. 2011; Hernández et al. 2014). This is particularly relevant for Mediterranean ecosystems that are characterised by frequent fires, which might contribute to explain the success of acacias such as *A. saligna* (Wilson et al. 2011) or *A. melanoxylon* (García et al. 2007). Moreover, under a climate change scenario, extreme and more frequent wildfires are expected in these ecosystems (IPCC 2013), which could effectively expand the distribution area of invasive acacias.

2.3 Symbiotic associations

As legumes, acacias are highly reliant on symbiotic associations with compatible microbes. In a new habitat, access to compatible rhizobia is a critical factor conditioning the invasive ability of legumes since mutualisms play a key role during their establishment (Parker 2001). Symbiotic promiscuity—low specificity for building up associations with compatible rhizobia—has been considered a characteristic trait of invasive legumes (Richardson et al. 2000b; Parker 2001). Invasive acacias can associate with a wide range of N-fixing bacteria (Lorenzo and Rodríguez-Echeverría, 2015). The invasive ability of acacias might be primarily determined by the capacity to form nodules profusely and more efficiently than native N-fixing legumes (Rodríguez-Echeverría et al. 2009, 2010). Acacias usually establish symbiotic relationships with the genus *Bradyrhizobium*, more specifically with *Bradyrhizobium japonicum* (Lafay and Burdon 2001; Rodríguez-Echeverría et al. 2007), in both native and non-native ranges (Birnbaum et al. 2012; Boudiaf et al. 2014). However, symbiotic interactions with new mutualists have also been reported in non-native ranges for Australian acacias (mainly *Bradyrhizobium* and *Rhizobium*, but also *Mesorhizobium*, *Ochrobactrum* and *Ensifer meliloti*) (Rodríguez-Echeverría et al. 2011; Birnbaum et al. 2012). For example, *A. saligna* may effectively associate with different rhizobial communities in non-native and native ranges (Birnbaum et al. 2012).

Nevertheless, *A. longifolia*, *A. dealbata* and *A. melanoxylon* preferentially associate with co-introduced symbionts in non-native ranges (Rodríguez-Echeverría et al. 2011; Lorenzo and Rodríguez-Echeverría, 2015), discarding symbiotic promiscuity as an invasive trait. In fact, Le Roux et al. (2016) have recently indicated that native and invasive legumes (*Acacia* within them) interact with distinct rhizobial lineages in South Africa. They found that instead of the classic vision of disrupting invasions, acacias and their symbionts form novel modules which are largely unconnected to highly modular native legume–rhizobium networks. Genetic analysis of symbiotic bacteria from root nodules of *A. saligna* from

Portugal indicated that obtained sequences mainly clustered with Australian sequences, suggesting the co-introduction of symbiotic partners (Crisostomo et al. 2013). Consequently, the rapid expansion and great nodulation ability of *A. longifolia* could enlarge the population and spread of the associated exotic *Bradyrhizobium* through the establishment of positive feedbacks (Rodríguez-Echeverría et al. 2009). The establishment of positive soil feedbacks has been also suggested when *A. dealbata* grows in previously invaded soils (Lorenzo and Rodríguez-Echeverría 2012; Rodríguez-Echeverría et al. 2013). This fact illustrates the ecological risk of the voluntary and involuntary introduction of exotic mutualistic microorganisms in reforestation projects. Invasion by acacia species may be favouring a second invasion by their associated exotic soil microbes. As a consequence, such synergistic interaction could accelerate impacts on ecosystems in the introduced ranges (Invasional meltdown hypothesis, Simberloff and Von Holle, 1999).

2.4 A clear role of allelopathy?

The release of allelochemicals by invasive plants has been postulated as a factor influencing the surrounding environment and favouring invasion (Inderjit et al. 2011). Allelopathy occurs because some IAPs bring novel chemicals that affect native species (Callaway and Aschehoug 2000). The allelopathic phenomenon has been broadly studied in *A. dealbata*. In the invaded ranges, extracts of *A. dealbata* containing natural or close to natural concentrations affected germination, seedling growth, net photosynthetic rate, respiration rate and biomass of agricultural and native understory plants (Carballeira and Reigosa 1999; Lorenzo et al. 2010b, 2011, 2012; Aguilera et al. 2015a) and functional diversity of soil microbes (Lorenzo et al. 2013a). Studying the release of allelochemicals along the different phenological stages of *A. dealbata*, Lorenzo et al. (2010c) found that allelopathic interactions were higher during the flowering period and depended on target species. A recent study showed that allelopathic effects mainly take place at root level, causing anomalous growth and morphology and leading to seedling mortality (Aguilera et al. 2015b). Interestingly, in vitro experiments with natural leachates obtained from adult *A. dealbata* plants increase the radicle length of its own seedlings, suggesting self-stimulation (Lorenzo et al. 2010c). However, the stimulatory effect disappeared when *A. dealbata* seedlings were grown on native soils (Lorenzo and Rodríguez-Echeverría 2012). In, volatile organic compounds (VOCs) released by *A. dealbata* flowers reduced germination and growth of its own seedlings (Souza-Alonso et al. 2014a). Despite the evidence of allelopathy under controlled conditions, the allelopathic effect was not detected at field scale, suggesting a negligible role of allelopathy during the invasive process of

A. dealbata, at least in the European range (Lorenzo et al. 2016a; Souza-Alonso et al. under review).

Much less information is available on the allelopathic potential of other acacias. Litter at different stages of decomposition and soils of *A. melanoxylon* have shown negative effects on the germination and growth of native plant species (González et al. 1995; Souto et al. 2001). Stem and bark aqueous extracts of *A. melanoxylon* reduced the growth of the aquatic plant *Lemna aequinoctialis* (Allan and Adkins 2007), whereas extracts from phyllodes and flowers of this species inhibited biometrical and physiological parameters of native and model species (Hussain et al. 2011a, b). Residues of *A. mearnsii* also showed a moderate allelopathic effect on the growth of dicotyledons and grasses (Schumann et al. 1995). Finally, Ens et al. (2009a, b) suggested that allelopathy plays an important role in ecological interactions of *A. longifolia* in their native range. However, these studies only constitute evidence of potential allelopathy since bioassays were conducted under controlled conditions. In fact, the effect of allelopathic compounds depends on bioassay conditions as the solvent, soil matrix or pH used and the presence/absence of soil microbes (Inderjit and van der Putten 2010; Lorenzo et al. 2016b). Therefore, experiments mimicking natural conditions are necessary to clearly identify the role of allelopathy in the invasive process. Otherwise, the allelopathic picture of the above-mentioned acacias will remain unclear and incomplete.

3 Effects on ecosystems

Invasive acacias affect both above- and belowground compartments as well as ecosystem services such as soil formation, water flow, nutrient cycling, wood or fibre production and recreation or educational opportunities that sustain human well-being (Le Maitre et al. 2011). The main characteristics of *Acacia* invasions are represented in Fig. 2. Nevertheless, the invasion of acacias presents geographical differences across Mediterranean regions.
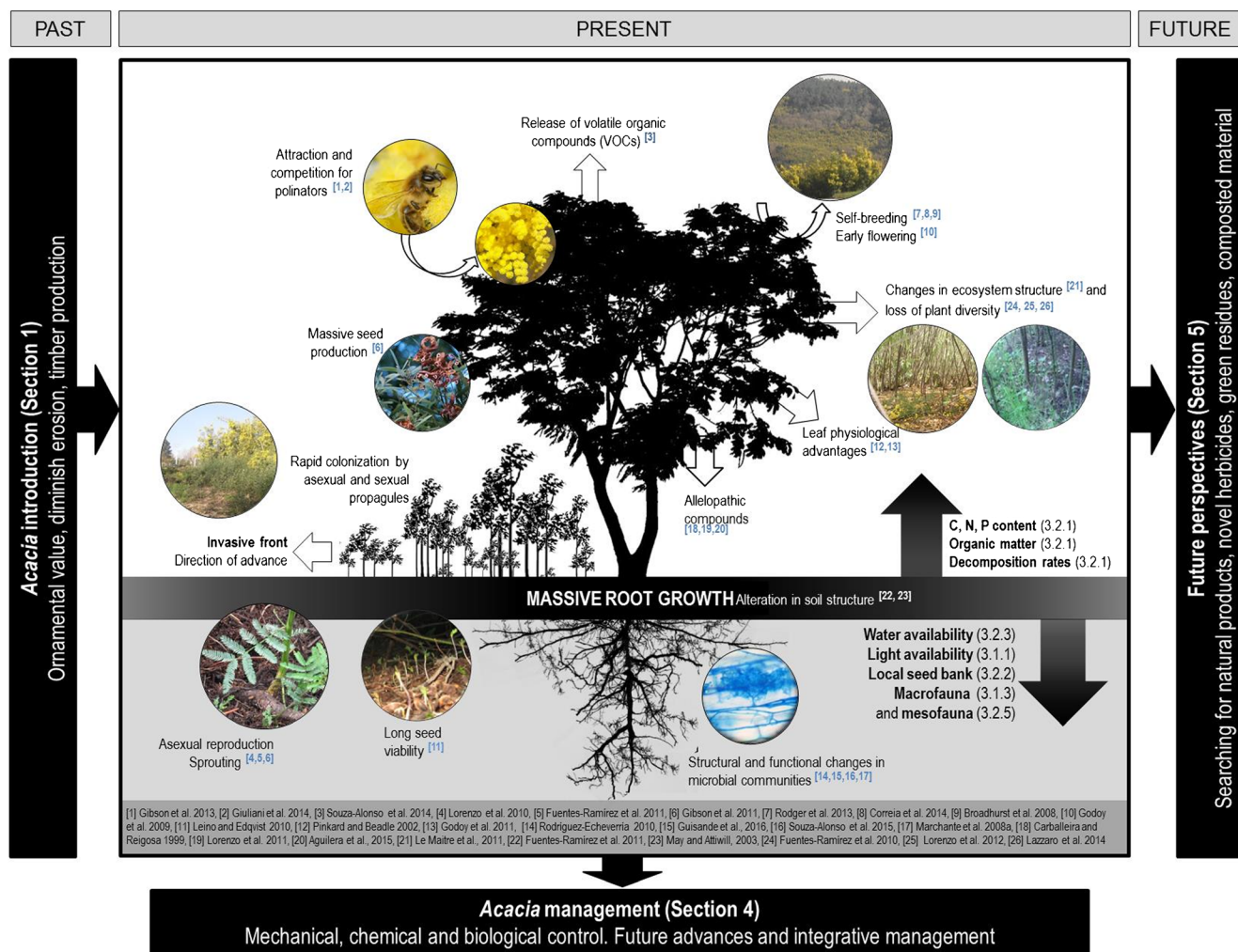
Fig. 2 Schematic representation of the main processes that take place under *Acacia* invasion and links to the main sections and references included in the manuscript

3.1 Aboveground effects

3.1.1 Structural changes

Invasive acacias create homogeneous and dense-vegetation formations (Le Maitre et al. 2011), which drastically decrease light availability for understory plants hindering their establishment (Lorenzo et al. 2010a; Rascher et al. 2011a; Lorenzo et al. 2016a). In fact, Fuentes-Ramírez et al. (2011) found a lower survival of light-demanding native forest species vs. shade-tolerant species under *A. dealbata*. The reduced light availability also leads to lower grass productivity through the reduction of specific leaf area index (LAI) thresholds (Gwate et al. 2016). However, *A. dealbata* did not reduce the light availability in broad-leaf native forests (González-Muñoz et al. 2012). This fact reveals that the influence of *A. dealbata* on light conditions is severe in native open canopies, but with slight effect in closed-canopy ecosystems.

Changes in the dominant tree species entail subsidiary consequences. Dense *Acacia* canopies lead to the accumulation of high quantity of biomass and litter, which increases the occurrence and intensity of fires in invaded ranges. Fires, in turn, stimulate the germination of acacia seeds and reduce the viability of native seeds favouring the invasive process (Richardson and Kluge 2008; Le Maitre et al. 2011). However, this fact has more ecological relevance in ecosystems without dominant species reliant on fire to germinate. In some Mediterranean areas, such as in central Chile, model projections predict the dispersion of *A. dealbata* only in the presence of fire when combined with browsing and/or cutting (Newton et al. 2011).

3.1.2 Plant biodiversity

In general, *Acacia* invasions significantly reduce plant cover, species richness and diversity (Holmes and Cowling 1997; Marchante et al. 2003; Tassin et al. 2009b; Fuentes-Ramírez et al. 2011; Lorenzo et al. 2012; Lazzaro et al. 2014). Biodiversity reduction due to *A. dealbata* invasion results in the replacement of native species by other natives or exotic plants (Fuentes-Ramírez et al., 2011; Lorenzo et al. 2012; Marchante et al. 2011b; González-Muñoz et al. 2012). In comparison with other invasive species, plantations of *A. saligna* have demonstrated a higher capacity to affect plant diversity (Manor et al. 2008). Surprisingly, *A. saligna* selectively increased the presence of ruderal grass species without reducing total richness (Del Vecchio et al. 2013). The identification of changes in plant species composition along invaded areas provides highly valuable information. Nonetheless, to our knowledge, whether modified native communities are accompanied by alterations in functional and phylogenetic diversity of invaded plant communities remains unknown.

3.1.3 Macrofauna

The presence of invasive acacias also modifies habitat suitability for animals and establishes novel ecological networks. Van der Colff et al. (2015) found a different trend of arthropod community composition between native and invaded areas by *A. mearnsii*; arthropods could be using exotic trees as a pathway to reach isolated habitats. In this sense, leaf N content is an important driver of arthropod population dynamics in *A. mearnsii* stands (Maoela et al. 2016a). Nevertheless, arthropod assemblages in the native community can be progressively recovered after the removal of the exotic (Maoela et al. 2016b). On the other hand, Eichhorn et al. (2011) indicated that the artificial damage induced to the leaves of *A. dealbata* activated the production of extra-floral nectaries. After damage, leaves were only visited by the invasive Argentine ant *Linepithema humile*, which could imply an interspecific positive feedback between invasive species. Moreover, larger animals are also affected by acacia invasions. The tree density of *A. saligna* stands, together with other factors such as urban density or vegetation structure, contributed to the decline of birds diversity (Dures and Cummings 2010) and species of small mammals (Manor et al. 2008), linking the decrease in biodiversity with a reduction in habitat quality or ecosystem integrity. Additionally, seeds of *A. mearnsii* are used as a nutrient source by the specialist primate *Cercopithecus albogularis labiatus*, altering its feeding behaviour and probably leading to consequences for *A. mearnsii* dispersion (Wimberger et al. 2017).

3.2 Belowground effects

3.2.1 Physicochemical changes and nutrient cycling

The rapid observation of the understory below the canopy of acacias indicates substantial changes in the structure of soil surface, linking *Acacia* invasion with the concept of *niche construction* (Day et al. 2003). The overwhelming surface root development of *Acacia* trees dominates and drastically transforms soil surface. *Acacia dealbata* creates a root net in the upper soil layer due to its extensive creeping rhizomatous system (Fuentes-Ramírez et al. 2011), reducing soil bulk density (May and Attiwill 2003). Similarly, *A. saligna* develops roots reaching 6 m during the first 4 years (Knight et al. 2002). Below the canopy, a thick layer of organic matter is progressively accumulated by the continuous litter fall (Marchante et al. 2004; Castro-Díez et al. 2012). Acacias provide litter with different C-sources composition that can affect nutrient cycling and decomposition, with possible ecological ramifications (Ens et al. 2009a). Nevertheless, decomposed plant material of *A. dealbata* did not produce significant changes in the functional and structural profile of soil microbial communities and soil chemical properties compared to the decomposition

of similar quantities of native plant material (Guisande et al. in preparation).

As N$_2$ fixers, acacias increase N (Marchante et al. 2008a; Lorenzo et al. 2010b; Souza-Alonso et al. 2014b) or NH$_4^+$ pools (Castro-Díez et al. 2012). *Acacia saligna* modifies N cycling through the production of higher amounts of litter, resulting in more N being returned to the soil and an increase in the availability of inorganic N (Yelenik et al. 2004). *Acacia longifolia* provides large quantities of N to the surrounding vegetation; however, at the same time, requires substantial amounts of P itself which creates a N/P imbalance at the community level (Ulm et al. 2016). Moreover, acacias substantially and progressively change C content in long-time invaded soils (Yelenik et al. 2004; Marchante et al. 2008a; Souza-Alonso et al. 2015). Other parameters, such as the content of organic matter or interchangeable P, were significantly increased by *A. dealbata* in soils from different ecosystems (Lorenzo et al. 2010b; Souza-Alonso et al. 2014b). However, Castro-Díez et al. (2012) found no differences in pH or organic matter after *A. dealbata* invasion. Souza-Alonso et al. (2014b) suggested that the variation in pH might be highly dependent on the studied ecosystem. *Acacia longifolia* drastically increased the content of C and N, C/N ratio, pH and litter in ecosystems with poor soils, such as sand dunes and coastal areas (Marchante et al. 2008a, c; Rascher et al. 2011a), resulting in differences in the catabolic diversity of microbial communities (Marchante et al. 2008c). Interestingly, these soil changes lead to a positive feedback between acacias and invaded soils. Soils previously invaded by *A. dealbata* favour the growth of its own seedlings and increase the mortality of the co-occurring native *Pinus pinaster* Aiton (Lorenzo and Rodríguez-Echeverría 2012; Rodríguez-Echeverría et al. 2013). This legacy effect—persistent changes in the long term—may continue even after acacia removal (Marchante et al. 2008b, 2011a).

3.2.2 Seed bank

The composition of the soil seed bank after acacia invasion is significantly modified by limiting or interrupting native propagule supply. Richness of native seeds was drastically decreased after the increase in *A. longifolia* density, while seeds of the invader were progressively accumulated (Fourie, 2008, Richardson and Kluge 2008; Le Maitre et al. 2011). Similarly, the diversity of the seed bank in understories invaded by *A. saligna* and *A. dealbata* was severely affected (Holmes and Cowling 1997; González-Muñoz et al. 2012), resulting in a diminution and homogenisation in the size of the native seed bank and higher percentages of exotic seeds in invaded ecosystems (Marchante et al. 2011b; González-Muñoz et al. 2012).

3.2.3 Water relationships

Water availability is often indicated as one of the main limiting factors of plant growth in Mediterranean areas (Claeys and Inzé 2013; Flexas et al. 2014). Across their range of introduction, invasive acacias are considered as water-consuming species, and their presence leads to a reduction in the quantity and quality of available water in soil and an increase in the evapotranspiration rate (Lorenzo and Rodríguez-Echeverría 2015). In the non-native range, water consumption by *A. melanoxylon* was higher than that measured for highly competitive species such as *Eucalyptus globulus* or *P. pinaster* (Jiménez et al. 2010). In South Africa, besides the use of groundwater, *A. dealbata* and *A. mearnsii* collected an important part of the estimated reduction of the mean annual runoff produced by all invasive plants (Le Maitre et al. 2000). This is particularly relevant in areas that present very low surface runoff, as in coastal arid regions. Novel *A. mearnsii* populations presented higher water losses compared to natives (Dye et al. 2001), whereas *A. longifolia* reduced the water flow on average by 26% in pine forests of coastal dunes in Portugal (Rascher et al. 2011b). At the same time, changes in hydrologic dynamics produced by *A. longifolia* were also associated with decreased C fixation rates of native trees (Rascher et al. 2011b). Interestingly, the high water consumption is generally considered a strategy for individual fast growth. Nevertheless, due to the ability of acacias to sprout, water consumption could be alternatively seen as a community-level strategy promoting the collective rather than individual plants in the long term (Werner et al. 2010).

Acacias can also influence the water availability for surrounding plant communities through other strategies at root level. High molecular weight alkanes exuded from roots by *A. longifolia* can induce water repellence, thereby reducing the accessible water for native seedlings (Ens et al. 2009b). However, under stressful conditions of limited water supply, *A. longifolia* revealed high drought sensitivity in terms of biomass and N-uptake efficiency, which was even more marked when plants grew with intra- or interspecific competition (Werner et al. 2010). Considering the evolutionary link that relates drought-tolerant xylem structure with the capacity to resist lower water potentials (Bhaskar and Ackerlyt1 2006), *A. mearnsii* showed lower water potential at 50% hydraulic conductivity loss (P$_{50}$) compared to native species, suggesting drought-tolerance (Crous et al. 2012). Field xylem water potentials also support that *A. mearnsii* has a significant advantage over some native species under drier conditions (Crous et al. 2012).

The removal of acacias might facilitate the replenishment of water for native vegetation, becoming a key factor to be considered in management operations, particularly in Mediterranean areas. In fact, removal of *A. mearnsii* and *A. longifolia* from riparian habitats increased the streamflow

(Prinsloo and Scott, 1999). Marais and Wannenburgh (2008) suggested that the removal of invasive acacias does not immediately imply water availability, but they consider it as an important part of a package of several actions to optimise water supply. Jovanovic et al. (2009) indicated that clearing lands invaded by *A. saligna*, besides the increase in water availability due to the reduction in evapotranspiration, may also reduce the contamination of groundwater by nitrate. Notwithstanding, to be realistic, changes in water regimes attributed to *Acacia* invasions or plantations should also include climatic conditions (rainfall patterns) as a potential source of variability (Rangan et al. 2010).

3.2.4 Soil microorganisms

Recent studies found substantial changes in soil microbial communities at structural and functional level produced by *Acacia* invasion (Marchante et al. 2008a, c; Lorenzo et al. 2010b, 2013a; Boudiaf et al. 2013; Souza-Alonso et al. 2014b, 2015). These changes are more pronounced in the long term or in heavily invaded areas and depend on the invaded ecosystem (Marchante et al. 2008a; Lorenzo and Rodríguez-Echeverría 2015). In addition, bacteria seemed to be more affected than fungi (Marchante et al. 2008a; Lorenzo and Rodríguez-Echeverría 2015).

Bacteria *Acacia* invasion affects both the structure and functional diversity of soil bacterial communities (Lorenzo and Rodríguez-Echeverría 2015). Particularly, *A. longifolia* and *A. dealbata* alter the structure of bacterial communities from dunes, grasslands and mixed forests (Marchante et al. 2008a, c; Lorenzo et al. 2010b), relating the duration of the invasion with the magnitude of the effect produced (Marchante et al. 2008a; Souza-Alonso et al. 2015). On the other hand, the functional catabolic diversity of soil bacteria also varies after the invasion by *A. longifolia*, *A. dealbata* and *A. mearnsii* (Marchante et al. 2008c; Boudiaf et al. 2013; Lorenzo et al. 2013a).

Fungi The effect of invasion on soil fungal communities was mainly studied in soils invaded by *A. dealbata*, which modifies the community structure of generalist fungi in pine forests and shrublands, but the effect depend on the studied ecosystem (Lorenzo et al. 2010b; Souza-Alonso et al. 2014b). Nevertheless, fungal communities seemed to evolve tolerance to invasion since they tended to return to the structure of pre-invaded community after long periods (>25 years) of invasion (Souza-Alonso et al. 2015). *Acacia* invasion also modified specific fungal groups such as arbuscular mycorrhizal fungi (AMF) and ectomycorrhizal fungi (EM). Structural changes in AMF communities caused by *A. dealbata* were accompanied by a reduced growth of the highly AMF-reliant plant *Plantago lanceolata* (Guisande-Collazo et al. 2016). However,

chemical compounds naturally released by *A. dealbata* did not affect the potential infectivity of AMF in different native soils (Lorenzo et al. 2013b). Similarly, *A. mearnsii* significantly altered the structure and composition of EM which, in consequence, produced a decrease in the early growth of the native tree *Quercus suber* L. (Boudiaf et al. 2013).

3.2.5 Mesofauna

The relationships between native plants and the community of decomposers can be also altered due to the presence of acacias. However, despite its fundamental role, studies addressing impacts of acacias on groups implicated in the breakdown of organic matter are scarce. Coetzee et al. (2007) found a significant reduction in richness, abundance and body size of arthropods (Coleoptera) in grasslands invaded by *A. dealbata* compared to non-invaded areas. Additionally, the presence and litter production of *A. mearnsii* in riparian habits altered the structure of invertebrate communities, reducing the abundance of some cobble-dwelling taxa but increasing particle-feeding mayflies and chironomids (Lowe et al. 2008). Below *A. melanoxylon* and *A. mearnsii* canopies, invertebrate richness was reduced compared to that under native species, and this reduction was higher at species level than at family or order level (Samways et al. 1996), indicating that changes in the dominant species has probably lower implications at functional level. Furthermore, qualitative changes in litter composition produced by *A. dealbata* and *A. longifolia* invasion result in poor nutrient material for terrestrial isopods—key components of macro-decomposer communities—leading to smaller individuals (Sousa et al. 1998).

4 Control and management

4.1 Recent advances in traditional control

Research on *Acacia* management started in South Africa, a pioneer country implementing management policies at national level. First organised efforts to control *A. dealbata*, *A. longifolia* or *A. mearnsii* were carried out mainly through the implementation of the Working for Water program (van Wilgen et al. 2011 and references therein). In general, the management of acacias is an expensive investment and long-time task due to the sprouting ability and their large and resilient seed banks (Richardson and Kluge 2008; Gaertner et al. 2012; van Wilgen et al. 2016).

Potential effective results have been achieved using triclopyr herbicide combined with cutting of *A. dealbata* individuals in a short-term strategy (Campbell and Kluge 1999; Souza-Alonso et al. 2013). Triclopyr was also effective to control *A. mearnsii* seedlings, even at low doses (Viljoen

and Stoltsz 2008). Herbicide combined with cutting was useful to reduce *A. saligna* in post-burning control. However, cutting *A. saligna* saplings below the coppicing point produced the best results (Krupek et al. 2016). In other cases, the knowledge of the best phenological stage to manage acacias improves the effectiveness of management actions. For instance, basal cuttings of young *A. mearnsii* individuals (≤7 years) should be preferably done in non-growing periods to diminish sprouting (Perrando and Corder 2006). On the other hand, similar management actions may yield different results at different locations due to the specific site conditions and life history traits. In South Africa, the felling and removing of *A. mearnsii* produced both positive and negative results, which could be related to local specific conditions (Blanchard and Holmes 2008). Nevertheless, results obtained during the last decades showed that the successful recovering of invaded areas by using traditional control methods is difficult to achieve due to the extension of invasion invaded areas (van Wilgen et al. 2012).

4.2 Biological control

The biological control of acacias started with the introduction of the bud-galling wasp, *Trichilogaster acaciaelongifoliae*, to control *A. longifolia* in South Africa (Dennill and Donnelly 1991). After several generations, the production of *A. longifolia* pods has been highly reduced. However, the effectiveness of the bud-galling agent was higher in areas with similar atmospheric conditions to native regions of the introduced wasp. In addition, a recent study found that populations of *A. longifolia* showing high genetic variability may differentially respond to the control agent in introduced ranges (Thompson et al. 2015), hampering the success of the biological control and compromising the reproducibility of this method. Similarly, the beetle *Melanterius ventralis* was introduced to feed on seeds of *A. longifolia*, producing seed mortality in a range from 15 to 79.5% (Donnelly and Hoffmann 2004). In Portugal, *T. acaciaelongifoliae* was recently introduced and tested on *A. longifolia* under controlled conditions with positive results (Marchante et al. 2011c). Subsequently, the European Commission (EC), after approval by the EFSA Panel on Plant Health (EFSA 2015; Jeger et al. 2016), authorised field tests that were conducted in late 2015 (Shaw et al. 2016). First reports indicated that *T. acaciaelongifoliae* successfully completed its life cycle in Portugal although the number of detected galls is currently low (Marchante et al. 2017). The flower-galling midge *Dasineura rubiformis* was also effectively introduced to control *A. mearnsii*, exclusively affecting its reproductive capacity (Impson et al. 2008, 2013). During the period of 1991–2005, the introduced rust-fungus *Uromycladium tepperianum* significantly affected *A. saligna* by reducing tree density (between 87 and 98%) and canopy mass, also increasing tree mortality (Wood and Morris 2007).

However, undesirable side effects of biological control may occur (Seymour and Veldtman 2010; Veldtman et al. 2011). In South Africa, the liberation of some control agents such as *T. acaciaelongifoliae*, *Dasineura dielsi* and *M. ventralis* unintentionally damaged the non-target *A. melanoxylon*, *A. longifolia* and *A. melanoxylon*, respectively (Dennill et al. 1993; Post et al. 2010; Donnelly and Hoffmann, 2004). This could be related to the low specificity of biocontrol agents that can lead to affinities for related species (Donnelly and Hoffmann, 2004). In fact, congeneric plats closely related to the target species are more susceptible to be also attacked than distantly related ones (Pemberton, 2000). In these cases, the side effect can be considered "positive" since other invasive congeners (all leading to acacia control) were targeted. Therefore, the use of biological control agents in Europe or North America to control acacias should have presumable low ecological risks due to the absence of native acacias. On the other hand, ecological effects of introduced agents are not completely addressed and unexpected consequences as ecological replacement, compensatory responses or food-web interactions may occur (Pearson and Callaway 2003). In fact, agents introduced to control *A. longifolia* and *A. saligna* in South Africa created complex food webs in the introduced range, similar to those in their native ranges (Veldtman et al. 2011). Main ecological pressures or inconsistencies derived from the introduction of novel agents were identified by Louda et al. (2003) as the susceptibility of related species, host specificity determined by physiological range, increase in the extinction risk of vulnerable species, or the infiltration in natural areas away from targeted agroecosystems.

4.3 Effective recovery of cleaned areas

Theory predicts that management programs are more effective if invaders are rapidly recognised and the time between the introduction and management is as short as possible (Simberloff et al. 2013; Luque et al. 2014; Kimball et al. 2015). The intensity of the required intervention for ecosystem recovery is proportional to the duration (i.e. density) of invasion (Holmes et al. 2000). Furthermore, the early detection of invasive plants also contributes to a cost-effective management. Economic costs of clearing dense invaded areas are 3–20 times higher than those necessary to manage scattered invaded areas (Marais and Wannenburgh 2008). In this sense, the current regulation of the European Commission on invasive species foresees three types of interventions: prevention, early detection and rapid eradication and management (EC 2014). However, the success of land restoration after acacia removal is uncertain because of the severe changes in soil physicochemical properties (Marchante et al. 2004, 2011a, b). The transformation of ecosystems invaded by acacias suggests that a return to pre-existing conditions is virtually

impossible. Therefore, the concept of *restoration* should be understood as a synonym of *recovery*.

After the removal of invasive acacias, the ecosystem recovery takes several years before soil nutrients and processes return to similar pre-invasion levels. In fact, the autonomous recovery potential of native vegetation after clearing of dense *Acacia* stands is certainly limited (Mostert et al. 2017). For example, the recuperation of native plant communities in coastal sand dunes is difficulted by the time elapsed from the introduction of *A. longifolia*. Thus, eradication efforts should be maintained in the long term to achieve positive results (Marchante et al. 2008b). To develop efficient recovery programs, secondary effects after the removal on invaders must be also considered. In this line, the enhanced content of N in invaded soils favours the settlement of grasses, forbs and other shrubs, but hinders ericoid or proteoid species (Gaertner et al. 2012). Additionally, the growth rates of the nitrophilous species *Ehrharta calycina* increased in stands where *A. saligna* was removed, suggesting that subsequent invasions by weeds may occur after clearing N_2-fixing alien species (Yelenik et al. 2004). Consequently, ecosystem recovery can be facilitated by the simultaneous removal of the N-rich litter layer, facilitating the germination of native species in the short term (Marchante et al. 2004, 2008b). Nevertheless, a field study assessing long-term consequences of *Acacia* removal found that the recovery of native vegetation in 15-year-old cleared sites was accompanied by a gradual improvement in soil nutrient levels (Ndou and Ruwanza 2016). Removal without an adequate planning of management can lead to the exposure of infertile subsoil vulnerable to erosion, even more in areas with slow rates of plant colonisation such as hill slopes (Van Der Waal et al. 2012). This fact also restricts the colonisation by indigenous species that could aid in the soil stabilisation (de Neergaard et al. 2005).

The maintenance of the native seed bank is fundamental to successfully recover ecosystems after *Acacia* invasion. Unsuccessful recovering of invaded ecosystems after acacia removal is frequently related to the lack of native seeds or propagule supply (Galatowitsch and Richardson 2005). If the native seed bank is severely depleted after plant invasion, autogenic recovery can be inhibited (Le Maitre et al. 2011). In fact, when the seed bank is exhausted or reaches critical values, the inclusion of native seeds in restoration programs could be essential to achieve pre-existing conditions. For example, the re-introduction of riparian species is required in highly transformed river basins to promote recovery and prevent re-invasion (Holmes et al. 2005). In addition, native species with low nutrient requirements and strong competitive ability that can outcompete invasive acacias at the early seedling stage are particularly valuable (Werner et al. 2010), which may facilitate ecosystem recovery.

At the same time, massive seed banks of acacias are difficult to manage after the removal of acacias (Richardson and Kluge 2008). In some cases, fire was used to manage the acacia seed bank in dense invaded stands (Krupek et al. 2016). The application of fire after tree removal reduces the content of N in soil, causes a mass germination of *Acacia* seeds and occasionally stimulates the indigenous seed bank, as in fire-prone ecosystems (Le Maitre et al. 2011). Nevertheless, fire has negative consequences, and prescribed burns are only recommended under specific circumstances, as steep slopes or inaccessible areas (Fill et al. 2017). In general, fire should be used judiciously, combined with other methods or discarded in situations where conservation of indigenous biological diversity is of central consideration (Richardson and Kluge 2008). Soil surface temperature can be modified without the use of fire. In the case of small invaded areas, the dormancy of *Acacia* seeds might be artificially removed through soil solarisation. For example, Cohen et al. (2008) achieved a complete exhaustion of buried seeds of *A. saligna* using polyethylene mulches to impede the photosynthetic process and produce hydrothermal stress.

However, active restoration actions are rarely implemented after clearing invaded areas, unless the cost/benefit ratios are deemed acceptable (Fill et al. 2017). Active restoration can be effective and even financially feasible when compared to passive restoration. The density of exotic tress generally determines whether the economic balance of restoration is positive or negative (Gaertner et al. 2012). There is increasing evidence that, in some cases, the restoration of invaded areas is feasible and can provide multiple social and economic benefits (Murcia et al. 2014).

4.4 Towards an integral management

Experience obtained in the management of acacias has shown that successful projects require clear and time-based goals, adequate resources and actual and in-kind support from the stakeholders (Forsyth et al. 2012). An improved management strategy based on recently developed frameworks (Kumschick et al. 2012, 2015; Blackburn et al. 2014; Hawkins et al. 2015) should focus on priority areas and species, assuming trade-offs between preserving biodiversity and avoiding the expansion of the invasion; otherwise, money allocated to control actions will be wasted (van Wilgen et al. 2016).

However, until now, management actions conducted in priority areas showed little progress in reducing total infestation (van Wilgen et al. 2012; Gwate et al. 2016). Even in South Africa where public funds were periodically invested and maintained to control invasive acacias, the economic resources were clearly insufficient to eradicate the invasive acacias (van Wilgen et al. 2012). Combining management techniques such as the integrated use of fire and active re-seeding of cleared areas with indigenous shrubs would substantially increase the effectiveness of ecosystem restoration (Fill et al. 2017). Profitable land uses, selective thinning of invasive

aboveground biomass or grazing could enhance multi-benefits in invaded landscapes (Seastedt et al. 2008; Gwate et al. 2016).

Spatiotemporal modelling approaches, such as individual-based models (IBMs), stochastic dynamic methodology (StDM), or species distribution models (SDMs) are being developed to anticipate *Acacia* invasions and manage their impacts in Mediterranean areas (Thompson et al. 2011; Santos et al. 2015). However, SDMs combined with phylogeographic approaches were not totally effective in predicting the occurrence of the two subspecies of *A. dealbata* (*A. dealbata* ssp. *dealbata* and spp. *subalpina*) in South Africa (Hirsch et al., 2017). Recently, hierarchical framework that combines SDMs, scenario analysis and cost analyses to improve the assessment of *Acacia* invasions at regional and local scales has also been developed (Vicente et al. 2016). In addition to previous approaches, impacts of acacias in a specific area can be initially assessed by using the generic impact scoring system (GISS), a novel and feasible tool to easily quantify ecosystem impacts (Nentwig et al. 2016).

In our opinion, the current vision of *Acacia* management by scientists is mainly focused on the ecological perspective, avoiding socioeconomic implications. Generally, management actions are carried out with public sources, resulting in an unavoidable necessity of social and scientific alliances. Public perception of IAPs is a key part in the assessment of management strategies, therefore providing a favourable social and political environment which is essential to achieve successful results. The engagement of public perception in management actions is more efficient and accepted by both parts (Panetta and Timmins 2004). In this sense, the use of inquiries is currently gaining interest as an informative and feedback tool in decision-making processes (Verbrugge et al. 2014; Liu and Cook 2016). Otherwise, eradication efforts are useless when administration and social actions do not pursue similar interests, suggesting that local communities need to be actively involved in the control of IAP and management programs (Mukwada and Manatsa 2017).

After several years of observation, we are also certain that socioeconomic aspects such as the forced human migration from rural to urban areas leads to land neglect and misuse and this movement is favouring the invasion by *Acacia*—and also other IAPs. Facilitating the settlement of population in rural areas would help to quickly identify and avoid the dispersal of acacia propagules, preserving rural native vegetation. In fact, increase access to land use for farming purposes could result in a greater concern, care and, ultimately, a better management of acacias (de Neergaard et al., 2005). Unfortunately, unworked or unprotected lands do not represent a significant value for the society. To us, government policies exclusively focused on the control of IAPs, but avoiding the problem of land misuse, cannot be totally effective.

Moreover, in many areas worldwide, the governmental actions to control acacias rely on workers that are seasonally recruited and do not necessarily return the following season (Fill et al. 2017). In other cases, as in the Working for Water program, the objective of maximising employment (reducing cost/person day) limits the effective monitoring and evaluation of outcomes due to poorly trained workforce (van Wilgen and Wannenburgh 2016). Alternatively, operational models that extend monitoring units throughout the year would lead to a better IAP management, saving economic funds (e.g., training costs) and ameliorating decision-making processes (Fill et al. 2017). In our point of view, an effective and sustainable control of acacias should include not only management actions and continuous monitoring, but also the maintenance of population in rural areas, thereby facilitating the surveillance and stability of ecosystems. Further actions to include the participation of society should also be a motivational challenge for those social agents involved in controlling IAPs (Le Maitre et al. 2011). Idealistically, in the current context of a changeable economic scenario and unsustainable consumption of resources, policies adopting long-term initiatives to ameliorate human life conditions, reorganising our concepts of human progress, sustainable society and land development, are required.

5 Future research and perspectives

Here to stay? Was a rhetorical question proposed by Richardson et al. (2011) exploring the human dimension—historical, scientific, social—of introduced acacias. In our opinion, *Acacia* invasions are far from being fully understood and foreseeable, becoming a challenging task for the next decades. In a context of climate change and land use alterations, Mediterranean ecosystems are under the pressure of new invasions by *Acacia* species. Wilson et al. (2011) recommended key topics of short- and long-term research to understand and manage potential invasiveness of invasive acacias, highlighting the importance of seed bank dynamics and seed dispersal, biogeographical comparisons to understand successful introductions, control and responsible actions (including public awareness). In this sense, emerging tools such as modelling, genomics, remote sensing and new imaging tools, the elaboration of improved ecological databases or the application and amelioration of allometric equations for biomass estimation based on larger forestry datasets will contribute to answer past and future questions regarding *Acacia* invasions. According to our experience, acacia stands should be considered as an *entity* instead of a group of individuals due to the massive vegetative reproduction. Thus, the clonality, physiological integration or resource allocation are topics that remain poorly understood for invasive acacias.

Acacias are catalogued as undesirable plants while, at the same time, their cultivation also provides profitable resources in different countries. It is therefore fundamental to determine

the trade-off between the commercial value and related environmental problems. To avoid the undesirable impacts without interfering with industry purposes, the implementation of sterile lineages of acacias is under investigation (Beck and Fossey 2007; Beck-Pay 2013). We also suggest that forest managers, industries or land owners that benefit from the cultivation of exotic acacias should be economically responsible for the problems derived from their plantations. Law reinforcement to unify forest regulations, especially among countries in the Mediterranean basin such as Spain, Portugal or Italy, is necessary to avoid further introduction of invasive acacias.

Current socioeconomic conditions are unstable in many countries, which imply that cost-effective management investments should be preferred instead of those which uniquely imply costs. Large management actions are probably unsustainable in the long term, whether they are entirely dependent on external funding (de Neergaard et al. 2005). In this line, we suggest that obtaining benefits of residues obtained from the management of acacias could alleviate the cost of the management. Therefore, we compiled several incipient research areas where acacias could be useful:

1. *Agriculture:* according to the directive on the sustainable use of pesticides proposed by the European Commission (2009/128/EC), the excessive use of synthetic herbicides should be reduced. In this sense, phytotoxicity compounds of invasive acacias could be used as a base to develop new bio-herbicides, bio-pesticides or phytotoxic mulches to control weeds or plagues in crops (Narwal 2010; Jabran et al. 2015). In fact, studies to identify the phytotoxic activity of chemical compounds from different *A. dealbata* material (Lorenzo et al. 2016b) and the use of green manures from *A. dealbata* and *A. longifolia* as bio-herbicides in agricultural soils (Souza-Alonso et al. under review) are currently in progress. Similar to other legume species (Narwal 2010), acacias pose nutrient-enriched leaves that could be used as fertilisers and a source of nutrients for crops. After full compost maturation, *A. longifolia* and *A. melanoxylon* provide agricultural amendments, biocomposts, with high organic matter content and low electrical conductivity (Brito et al. 2013, 2015). Composting residues of *A. dealbata* with sewage sludge also improves soil biochemical and chemical properties (Tejada et al. 2014). The use of acacia residues can be included into the current idea of changing towards a green economy, in the framework of the bioeconomy strategy (H2020 Program).

2. *Industry:* the high polysaccharide content of *A. dealbata* is a valuable resource for biorefineries, providing a way of upgrading underused renewable feedstocks (Yañez et al. 2009, 2013). New cationic polymeric coagulants for water and different types of industrial effluent treatments were synthesised with tannin extracted from *A. mearnsii*

(Beltrán-Heredia et al. 2010, Sánchez-Martín et al. 2012; Soares et al. 2012), having also potential as phytoextractor in the remediation of heavy metal contaminated biosolids (Mok et al. 2013). Similarly, Kumari and Ravindhranath (2012) successfully employed *A. melanoxylon* as biosorbent in the extraction of Al^{+3} ions from waste waters collected from industrial effluents and polluted lakes. In addition, extracts from *A. mearnsii* showed positive results to control blue algal blooms (Zhou et al. 2012).

3. *Health purposes:* acacias can also be a chemistry source of chemical components with medical and health purposes. In example, bark of *A. mearnsii* is traditionally used in the treatment of stomach diseases (Wintola et al. 2017). Crude extracts from this species also exhibited significant antimicrobial activity, becoming a potential source of bioactive compounds (Olajuyigbe and Afolayan 2012). Phenolic, flavonoid and alkaloid contents of raw extracts from *A. dealbata* and *A. melanoxylon* showed stronger antioxidant activities (Luis et al. 2012). Preliminary results also indicate that water-soluble compounds present in extracts of *A. melanoxylon* exhibited anthelmintic activity against larval development of horse parasites (Payne et al. 2013). Acacia honey induces the expression of cytokines and a metalloproteinase that degrades collagen IV involved in the disorganisation of basal membrane during the re-epithelialisation process of wounds (Burlando and Cornara 2013).

4. *Cosmetics*: Absolute oils from *A. dealbata* have been used in cosmetic industries, especially in the production of perfumes, due to the presence of odorant compounds (Perriot et al. 2010).

6 Conclusions

Substantial efforts have been carried out during the last years to address the consequences of the invasion of Mediterranean ecosystems by acacias. Nowadays, having left behind the consideration of emerging threats, acacias are recognised as severe menaces to Mediterranean ecosystems and the reinforcement of transnational regulations, together with the development of crossing-information platforms, seems crucial to prevent novel *Acacia* introductions. Under a future scenario of climate change, these ecosystems are expected to be largely occupied by invasive acacias due to their increased growth under higher CO_2 conditions, seed production and fire resistance. Changes in hydrological dynamics by acacia invasions may also exacerbate droughts in Mediterranean areas under expected extreme climatic events.

Invasions by acacias usually lead to changes in ecosystem services as water and fire regimes, reduction in plant biodiversity and alteration in soil physicochemical properties and

function. Modified soil microbial communities may have negative implications for nutrient cycling, ecosystem processes and native vegetation that rely on them, which, in turn, might favour acacia invasion and increase the vulnerability of affected ecosystems. In terms of the assessment of native plant communities, a deeper knowledge of the functional and phylogenetic diversity, rather than the use of classic diversity indices, should be considered to evaluate the extent of the ecological impacts produced. Further work is also needed to elucidate the proportion of sexual vs. vegetative reproduction during the invasion process to design adequate control strategies.

We consider that the management of acacias should be focused on prioritising the preservation of non-invaded habitats and the identification of areas with potential to host invasive acacias. Risk assessment studies, based on recently developed frameworks and more focused on forecasting and preventing future introductions rather than evaluate changes in already invaded areas, are also desirable. It is also time to communicate and to engage social, politician and stakeholder perceptions to provide integrative, sustainable and adapted solutions to *Acacia* invasion, since high economic investments do not necessarily assure the success in the control of *Acacia* invasions. The search of potential uses for acacia residues could possibly bring solutions to partially alleviate the economic resources allocated to their management and, at the same time, reduce the extension of invasive populations. Therefore, applied research on profitable uses for acacia residues seems to be highly relevant in the future.

After two centuries of introduction, rapid evolutionary processes could be occurring and should be an interesting point of future works. Ecologists and evolutionary biologists are at the forefront of a model group, with challenging research possibilities. In the same line, novel relationships between plant pollinators, plant-seed dispersers or plant herbivores and acacias can produce novel ecological interactions that could alter or displace well-established ecological networks. In this sense, the rhetorical question raised 6 years ago *here to stay?* should be currently transformed—as the title of our review indicates—into an affirmative sentence. The emerging assumption that the complete eradication of acacias seems, in some cases, unfeasible provides a new context in which the study of the ecological role of *Acacia* formations—as novel ecosystems—emerges relevant.

Compliance with ethical standards

References

Afonso C (2012) Plant-soil feedback and invasion by Australian acacias. Master thesis, Universidade de Coimbra

Aguilera N, Becerra J, Guedes LM, Villaseñor-Parada C, González L, Hernández V (2015a) Allelopathic effect of the invasive *Acacia dealbata* Link (Fabaceae) on two native plant species in south-central Chile. Gayana Bot 72:231–239

Aguilera N, Guedes LM, Becerra J, Baeza C, Hernández V (2015b) Morphological effects at radicle level by direct contact of invasive *Acacia dealbata* Link. Flora 215:54–59

Allan SM, Adkins SW (2007) The effect of medicinal plant extracts on growth of *Lemna aequinoctialis*. Allelopathy J 19:267–274

Beck SL, Fossey A (2007) Gamma irradiation induces sterility or seedlessness in black wattle (*Acacia mearnsii*). Seed Sci Technol 35:351–359

Beck-Pay SL (2012) The effect of temperature and relative humidity on *Acacia mearnsii* polyad viability and pollen tube development. S Afr J Bot 83:165–171

Beck-Pay SL (2013) Confirmation of cytotype stability in autotetraploid black wattle (*Acacia mearnsii*) trees using flow cytometry and size differences of the reproductive gametes. South Forests 75:1–6

Beltrán-Heredia J, Sánchez-Martín J, Gómez-Muñoz MC (2010) New coagulant agents from tannin extracts: preliminary optimisation studies. Chem Eng J 162:1019–1025

Bhaskar R, Ackerly DD (2006) Ecological relevance of minimum seasonal water potentials. Physiol Plant 127:353–359.

Birnbaum C, Barrett LG, Thrall PH, Leishman MR (2012) Mutualisms are not constraining cross-continental invasion success of *Acacia* species within Australia. Divers Distrib 18:962–976

Blackburn TM, Essl F, Evans T, Hulme PE, Jeschke JM, Kühn I et al (2014) A unified classification of alien species based on the magnitude of their environmental impacts. PLoS Biol 12:e1001850

Blanchard R, Holmes PM (2008) Riparian vegetation recovery after invasive alien tree clearance in the Fynbos biome. S Afr J Bot 74:421–431

Boudiaf I, Baudoin E, Sanguin H, Beddiar A, Thioulouse J, Galiana A et al (2013) The exotic legume tree species *Acacia mearnsii* alters microbial soil functionalities and the early development of a native tree species *Quercus suber*, in North Africa. Soil Biol Biochem 65:172–179

Boudiaf I, Le Roux C, Baudoin E, Galiana A, Beddiar A, Prin Y, Duponnois R (2014) Soil *Bradyrhizobium* population response to invasion of a natural *Quercus suber* forest by the introduced nitrogen-fixing tree *Acacia mearnsii* in El Kala National Park, Algeria. Soil Biol Biochem 70:162–165

Brito LM, Saldanha J, Mourão I, Nestler H (2013) Composting of *Acacia longifolia* and *Acacia melanoxylon* invasive species. Acta Hortic 1013:211–216

Brito LM, Reis M, Mourão I, Coutinho J (2015) Use of acacia waste compost as an alternative component for horticultural substrates. Commun Soil Sci Plant Anal 46:1814–1826

Broadhurst LM, Young AG, Forrester R (2008) Genetic and demographic responses of fragmented *Acacia dealbata* (Mimosaceae) populations in southeastern Australia. Biol Conserv 141:2843–2856

Buist ML (2003) Comparative ecology and conservation biology of two critically endangered acacias (*Acacia lobulata* and *A. sciophanes*) and two common, widespread relatives (*Acacia verricula* and *A. anfractuosa*) from the south-west of Western Australia. PhD thesis, The University of Western Australia

Burlando B, Cornara L (2013) Honey in dermatology and skin care: a review. J Cosmetic Dermatol 12:306–313

Callaway RM, Aschehoug ET (2000) Invasive plants versus their new and old neighbors: a mechanism for exotic invasion. Science 290:521–523

Campbell PL, Kluge RL (1999) Development of integrated control strategies for wattle. 1. Utilization of wattle, control of stumps and rehabilitation with pastures. S Afr J Plant Soil 16:24–30

Carballeira A, Reigosa MJ (1999) Effects of natural leachates of *Acacia dealbata* Link in Galicia (NW Spain). Bot Bull Acad Sinica 40:87–92

Carruthers J, Robin L, Hattingh JP, Kull CA, Rangan H, van Wilgen BW (2011) A native at home and abroad: the history, politics, ethics and aesthetics of acacias. Divers Distrib 17:810–821

Castro-Díez P, Godoy O, Saldaña A, Richardson DM (2011) Predicting invasiveness of Australian acacias on the basis of their native climatic affinities, life-history traits and human use. Divers Distrib 17:934–945

Castro-Díez P, Fierro-Brunnenmeister N, González-Muñoz N, Gallardo A (2012) Effects of exotic and native tree leaf litter on soil properties of two contrasting sites in the Iberian Peninsula. Plant Soil 350:179–191

Celesti-Grapow L, Bassi L, Brundu G, Camarda I, Carli E, D'Auria G, del Guacchio E, Domina G, Ferretti G, Foggi B et al (2016) Plant invasions on small Mediterranean islands: an overview. Plant Biosyst 150:1119–1133

Claeys H, Inzé D (2013) The agony of choice: how plants balance growth and survival under water-limiting conditions. Plant Physiol 162: 1768–1779

Coetzee BWT, van Rensburg BJ, Robertson MP (2007) Invasion of grasslands by silver wattle, *Acacia dealbata* (Mimosaceae), alters beetle (Coleoptera) assemblage structure. Afr Entomol 15:328–339

Cohen O, Riov J, Katan J, Gamliel A, Bar P (2008) Reducing persistent seed banks of invasive plants by soil solarization-the case of *Acacia saligna*. Weed Sci 56:860–865

Correia M, Castro S, Ferrero V, Crisóstomo JA, Rodríguez-Echeverría S (2014) Reproductive biology and success of invasive Australian acacias in Portugal. Bot J Linn Soc 174:574–588

Correia M, Montesinos D, French K, Rodríguez-Echeverría S (2016) Evidence for enemy release and increased seed production and size for two invasive Australian acacias. J Ecol 104:1391–1399

Crisostomo JA, Rodríguez-Echeverría S, Freitas H (2013) Co-introduction of exotic rhizobia to the rhizosphere of the invasive legume *Acacia saligna*, an intercontinental study. Appl Soil Ecol 64:118–126

Crous CJ, Jacobs SM, Esler KJ (2012) Drought-tolerance of an invasive alien tree, *Acacia mearnsii* and two native competitors in fynbos riparian ecotones. Biol Invasions 14:619–631

Daws MI, Davies J, Vaes E, van Gelder R, Pritchard HW (2007) Two-hundred-year seed survival of *Leucospermum* and two other woody species from the Cape floristic region, South Africa. Seed Sci Res 17:73–80

de Neergaard A, Saarnak C, Hill T, Khanyile M, Berzosa AM, Birch-Thomsen T (2005) Australian wattle species in the Drakensberg region of South Africa. An invasive alien or a natural resource? Agric Syst 85:216–233

Del Vecchio S, Acosta A, Stanisci A (2013) The impact of *Acacia saligna* invasion on Italian coastal dune EC habitats. CR Biol 336:364–369

Dennill GB, Donnelly D (1991) Biological control of *Acacia longifolia* and related weed species (Fabaceae) in South Africa. Agric Ecosyst Environ 37:115–135

Dennill GB, Donnelly D, Chown SL (1993) Expansion of host-plant range of a biocontrol agent *Trichilogaster acaciaelongifoliae* (Pteromalidae) released against the weed *Acacia longifolia* in South Africa. Agric Ecosyst Environ 43:1–10

Donnelly D, Hoffmann JH (2004) Utilization of an unpredictable food source by *Melanterius ventralis*, a seed-feeding biological control agent of *Acacia longifolia* in South Africa. BioControl 49:225–235

Dures SG, Cumming GS (2010) The confounding influence of homogenising invasive species in a globally endangered and largely urban biome: does habitat quality dominate avian biodiversity? Biol Conserv 143:768–777

Dye P, Moses G, Vilakazi P, Ndlela R, Royappen M (2001) Comparative water use of wattle thickets and indigenous plant communities at riparian sites in the western Cape and KwaZulu-Natal. Water SA 27:529–538

European Commission (2014) Regulation (EU) No 1143/2014. of the European Parliament and of the Council of 22 October 2014 on the prevention and management of the introduction and spread of invasive alien species. Off J Europ Un 317:35–55.

EFSA PLH Panel (2015) Risk to plant health in the EU territory of the intentional release of the bud-galling wasp *Trichilogaster acaciaelongifoliae* for the control of the invasive alien plant *Acacia longifolia*. EFSA J 13:4079

Eichhorn MP, Ratliffe LC, Pollard KM (2011) Attraction of ants by an invasive acacia. Insect Conserv Diver 4:235–238

Eilu G, Obua J (2005) Tree condition and natural regeneration in disturbed sites of Bwindi Impenetrable Forest National Park, southwestern Uganda. Trop Ecol 46:99–112

Ens EJ, French K, Bremner JB (2009a) Evidence for allelopathy as a mechanism of community composition change by an invasive exotic shrub *Chrysanthemoides monilifera* spp. *rotundata*. Plant Soil 316:125–137

Ens EJ, Bremner JB, French K, Korth J (2009b) Identification of volatile compounds released by roots of an invasive plant, bitou bush (*Chrysanthemoides monilifera* spp. *rotundata*), and their inhibition of native seedling growth. Biol Invasions 11:275–287

Fill JM, Forsyth GG, Kritzinger-Klopper S, Le Maitre DC, van Wilgen BW (2017) An assessment of the effectiveness of a long-term ecosystem restoration project in a Fynbos shrubland catchment in South Africa. J Environ Manag 185:1–10

Flexas J, Diaz-Espejo A, Gago J, Gallé A, Galmés J, Gulías J, Medrano H (2014) Photosynthetic limitations in Mediterranean plants: a review. Environ Exp Bot 103:12–23

Forsyth GG, Le Maitre DC, O'Farrell PJ, Van Wilgen BW (2012) The prioritisation of invasive alien plant control projects using a multi-criteria decision model informed by stakeholder input and spatial data. J Environ Manag 103:51–57

Fourie S (2008) Composition of the soil seed bank in alien-invaded grassy fynbos: potential for recovery after clearing. S Afr J Bot 74:445–453

French K, Major RE (2001) Effect of an exotic acacia (Fabaceae) on ant assemblages in South African fynbos. Austral Ecol 26:303–310

Fuentes-Ramírez A, Pauchard A, Cavieres LA, García RA (2011) Survival and growth of *Acacia dealbata* vs. native trees across an invasion front in south-central Chile. For Ecol Manag 261:1003–1009

Gaertner M, Nottebrock H, Fourie H, Privett SDJ, Richardson DM (2012) Plant invasions, restoration, and economics: perspectives from South African fynbos. Perspect Plant Ecol Evol Syst 14:341–353

Galatowitsch S, Richardson DM (2005) Riparian scrub recovery after clearing of invasive alien trees in headwater streams of the Western Cape, South Africa. Biol Conserv 122:509–521

Gallagher RV, Leishman MR, Miller JT, Hui C, Richardson DM, Suda J, Trávníček P (2011) Invasiveness in introduced Australian acacias: the role of species traits and genome size. Divers Distrib 17:884–897

García RA, Pauchard A, Peña E (2007) Seed bank, regeneration and growth of *Teline monspessulana* (L.) K. Koch after a forest fire. Gayana Bot 64:201–210

George N, Byrne M, Yan G (2008) Mixed mating with preferential outcrossing in *Acacia saligna* (Labill.) H. Wendl.(Leguminosae: Mimosoideae). Silvae Gen 57:139–145

Gibson MR, Richardson DM, Marchante E, Marchante H, Rodger JG, Stone GN et al (2011) Reproductive biology of Australian acacias: important mediator of invasiveness? Divers Distrib 17:911–933

Gibson MR, Pauw A, Richardson DM (2013) Decreased insect visitation to a native species caused by an invasive tree in the Cape floristic region. Biol Conserv 157:196–203

Godoy O, de Lemos-Filho JP, Valladares F (2011) Invasive species can handle higher leaf temperature under water stress than Mediterranean natives. Environ Exp Bot 71:207–214

González L, Souto XC, Reigosa MJ (1995) Allelopathic effects of *Acacia melanoxylon* R. Br. Phyllodes during their decomposition. For Ecol Manag 77:53–63

González-Muñoz N, Costa-Tenorio M, Espigares T (2012) Invasion of alien *Acacia dealbata* on Spanish *Quercus robur* forests: impact on soils and vegetation. For Ecol Manag 269:214–221

González-Muñoz N, Linares JC, Castro-Díez P, Sass-Klaassen U (2014) Predicting climate change impacts on native and invasive tree species using radial growth and twenty-first century climate scenarios. Eur J For Res 133(6):1073–1086

Goodwin BJ, McAllister AJ, Fahrig L (1999) Predicting invasiveness of plant species based on biological information. Conserv Biol 13:422–426

Griffin AR, Midgley SJ, Bush D, Cunningham PJ, Rinaudo AT (2011) Global uses of Australian acacias–recent trends and future prospects. Divers Distrib 17:837–847

Grotkopp E, Rejmánek M, Sanderson MJ, Rost TL (2004) Evolution of genome size in pines (*Pinus*) and its life history correlates: supertree analyses. Evolution 58:1705–1729

Guisande-Collazo A, González L, Souza-Alonso P (2016) Impact of an invasive N₂-fixing tree on arbuscular mycorrhizal fungi and development of native species. AoB Plants 8:plw018

Gwate O, Mantel SK, Finca A, Gibson LA, Munch Z, Palmer AR (2016) Exploring the invasion of rangelands by *Acacia mearnsii* (black wattle): biophysical characteristics and management implications. Afr J Range Forage Sci 33:265–273

Harris CJ, Dormontt EE, Le Roux JJ, Lowe A, Leishman MR (2012) No consistent association between changes in genetic diversity and adaptive responses of Australian acacias in novel ranges. Evol Ecol 26:1345–1360

Hawkins CL, Bacher S, Essl F, Hulme PE, Jeschke JM, Kühn I et al (2015) Framework and guidelines for implementing the proposed IUCN environmental impact classification for alien taxa (EICAT). Divers Distrib 21:1360–1363

Hernández L, Martínez-Fernández J, Cañellas I, de la Cueva AV (2014) Assessing spatio-temporal rates, patterns and determinants of biological invasions in forest ecosystems. The case of *Acacia* species in NW Spain. For Ecol Manag 329:206–213

Higgins SI, Richardson DM, Cowling RM, Trinder-Smith TH (1999) Predicting the landscape-scale distribution of alien plants and their threat to plant diversity. Conserv Biol 13:303–313

Higgins SI, Richardson DM, Cowling RM (2001) Validation of a spatial simulation model of a spreading alien plant population. J Appl Ecol 38:571–584

Hirsch H, Gallien L, Impson FA, Kleinjan C, Richardson DM, Le Roux JJ (2017) Unresolved native range taxonomy complicates inferences in invasion ecology: *Acacia dealbata* Link as an example. Biol Invasions 19:1715–1722

Holmes PM, Cowling RM (1997) Diversity, composition and guild structure relationships between soil-stored seed banks and mature vegetation in alien plant-invaded south African fynbos shrublands. Plant Ecol 133:107–122

Holmes PM, Richardson DM, Wilgen BW, Gelderblom C (2000) Recovery of South African fynbos vegetation following alien woody plant clearing and fire: implications for restoration. Aust Ecol 25:631–639

Holmes PM, Richardson DM, Esler KJ, Witkowski ETF, Fourie S (2005) A decision-making framework for restoring riparian zones degraded by invasive alien plants in South Africa. S Afr J Sci 101:553–564

Hussain MI, Gonzalez L, Reigosa MJ (2011a) Allelopathic potential of *Acacia melanoxylon* on the germination and root growth of native species. Weed Biol Manage 11:18–28

Hussain MI, González L, Souto C, Reigosa MJ (2011b) Ecophysiological responses of three native herbs to phytotoxic potential of invasive *Acacia melanoxylon* R. Br. Agrofor Syst 83:149–166

Impson FAC, Kleinjan CA, Hoffmann JH, Post JA (2008) *Dasineura rubiformis* (Diptera: Cecidomyiidae), a new biological control agent for *Acacia mearnsii* in South Africa. S Afr J Sci 104:247–249

Impson FA, Post JA, Hoffmann JH (2013) Impact of the flower-galling midge *Dasineura rubiformis* Kolesik, on the growth of its host plant *Acacia mearnsii* De Wild, in South Africa. S Afr J Bot 87:118–121

Inderjit, van der Putten WH (2010) Impacts of soil microbial communities on exotic plant invasions. Trends Ecol Evol 25:512–519

Inderjit, Wardle DA, Karban R, Callaway RM (2011) The ecosystem and evolutionary contexts of allelopathy. Trend Ecol Evol 26:655–662

IPCC (2013) Climate change 2013, the physical science basis. Contribution of working group I to the fifth assessment report of the intergovernmental panel on climate change. Cambridge University Press, Cambridge

Jabran K, Mahajan G, Sardana V, Chauhan BS (2015) Allelopathy for weed control in agricultural systems. Crop Prot 72:57–65

Jeger MJ, Pautasso M, Stancanelli G, Vos S (2016) The EFSA assessment of *Trichilogaster acaciaelongifoliae* as biocontrol agent of the invasive alien plant *Acacia longifolia*: a new area of activity for the EFSA Plant Health Panel? EPPO Bulletin 46:270–274

Jiménez E, Vega JA, Pérez-Gorostiaga P, Fonturbel T, Fernández C (2010) Evaluation of sap flow density of *Acacia melanoxylon* R. Br. (blackwood) trees in overstocked stands in north-western Iberian Peninsula. Eur J For Res 129:61–72

Jovanovic NZ, Israel S, Tredoux G, Soltau L, Le Maitre D, Rusinga F et al (2009) Nitrogen dynamics in land cleared of alien vegetation (*Acacia saligna*) and impacts on groundwater at Riverlands nature reserve (western Cape, South Africa). Water SA 35:37–44

Kimball S, Lulow M, Sorenson Q, Balazs K, Fang YC, Davis SJ et al (2015) Cost-effective ecological restoration. Restor Ecol 23:800–810

Knight A, Blott K, Portelli M, Hignett C (2002) Use of tree and shrub belts to control leakage in three dryland cropping environments. Aust J Agric Res 53:571–586

Krupek A, Gaertner M, Holmes PM, Esler KJ (2016) Assessment of post-burn removal methods for *Acacia saligna* in Cape flats sand fynbos, with consideration of indigenous plant recovery. S Afr J Bot 105:211–217

Kubešová M, Moravcova L, Suda J, Jarošík V, Pyšek P (2010) Naturalized plants have smaller genomes than their non-invading relatives: a flow cytometric analysis of the Czech alien flora. Preslia 82:81–96

Kulkarni MG, Sparg SG, Van Staden J (2007) Germination and post-germination response of *Acacia* seeds to smoke-water and butenolide, a smoke-derived compound. J Arid Environ 69:177–187

Kull CA, Tassin J, Rangan H (2007) Multifunctional, scrubby, and invasive forests? Wattles in the highlands of Madagascar. Mt Res Dev 27:224–231

Kull CA, Shackleton CM, Cunningham PJ, Ducatillon C, Dufour-Dror JM, Esler KJ, Zylstra MJ (2011) Adoption, use and perception of Australian acacias around the world. Divers Distrib 17:822–836

Kumari AA, Ravindhranath K (2012) Extraction of aluminium (III) ions from polluted waters using bio-sorbents derived from *Acacia melanoxylon* and *Eichhornia crassipes* plants. J Chem Pharmaceut Res 4:2836–2849

Kumschick S, Bacher S, Dawson W, Heikkilä J, Sendek A, Pluess T et al (2012) A conceptual framework for prioritization of invasive alien species for management according to their impact. NeoBiota 15:69–100

Kumschick S, Bacher S, Evans T, Marková Z, Pergl J, Pyšek P et al (2015) Comparing impacts of alien plants and animals in Europe using a standard scoring system. J Appl Ecol 52:552–561

Kyalangalilwa B, Boatwright JS, Daru BH, Maurin O, Bank M (2013) Phylogenetic position and revised classification of *Acacia sl* (Fabaceae: Mimosoideae) in Africa, including new combinations in *Vachellia* and *Senegalia*. Bot J Linn Soc 172:500–523

Lafay B, Burdon JJ (2001) Small-subunit rRNA genotyping of rhizobia nodulating Australian *Acacia* spp. Appl Environ Microbiol 67:396–402

Day RL, Laland KN, Odling-Smee FJ (2003) Rethinking adaptation: the niche-construction perspective. Perspect Biol Med 46:80–95

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. Sci Total Environ 497:491–498

Le Maitre DC, Versfeld DB, Chapman RA (2000) Impact of invading alien plants on surface water resources in South Africa: a preliminary assessment. Water Research Commission 26:397–408

Le Maitre DC, Gaertner M, Marchante E, Ens EJ, Holmes PM, Pauchard A, Richardson DM (2011) Impacts of invasive Australian acacias: implications for management and restoration. Divers Distrib 17:1015–1029

Le Roux JJ, Mavengere NR, Ellis AG (2016) The structure of legume–rhizobium interaction networks and their response to tree invasions. AoB Plants 8:plw038

Leino MW, Edqvist J (2010) Germination of 151-year old *Acacia* spp. seeds. Genet Resour Crop Evol 57:741–746

Lewis SL, Maslin MA (2015) Defining the Anthropocene. Nature 171:171–180

Liu S, Cook D (2016) Eradicate, contain, or live with it? Collaborating with stakeholders to evaluate responses to invasive species. Food Secur 8:49–59

Lorenzo P, Gonzalez L, Reigosa MJ (2010a) The genus *Acacia* as invader: the characteristic case of *Acacia dealbata* Link in Europe. Ann For Sci 67:1–11

Lorenzo P, Rodríguez-Echeverría S, González L, Freitas H (2010b) Effect of invasive *Acacia dealbata* Link on soil microorganisms as determined by PCR-DGGE. Appl Soil Ecol 44:245–251

Lorenzo P, Pazos-Malvido E, Reigosa MJ, González L (2010c) Differential responses to allelopathic compounds released by the invasive *Acacia dealbata* Link (Mimosaceae) indicate stimulation of its own seed. Aust J Bot 58:546–553

Lorenzo P, Palomera-Pérez A, Reigosa MJ, González L (2011) Allelopathic interference of invasive *Acacia dealbata* Link on the physiological parameters of native understory species. Plant Ecol 212:403–412

Lorenzo P, Pazos-Malvido E, Rubido-Bará M, Reigosa MJ, González L (2012) Invasion by the leguminous tree *Acacia dealbata* (Mimosaceae) reduces the native understorey plant species in different communities. Aust J Bot 60:669–675

Lorenzo P, Rodríguez-Echeverría S (2012) Influence of soil microorganisms, allelopathy and soil origin on the establishment of the invasive *Acacia dealbata*. Plant Ecol Divers 5:67–73

Lorenzo P, Pereira CS, Rodríguez-Echeverría S (2013a) Differential impact on soil microbes of allelopathic compounds released by the invasive *Acacia dealbata* Link. Soil Biol Biochem 57:156–163

Lorenzo P, Rodríguez-Echeverría S, Freitas H (2013b) No allelopathic effect of the invader *Acacia dealbata* on the potential infectivity of arbuscular mycorrhizal fungi from native soils. Europ J Soil Biol 58:42–44

Lorenzo P, Rodríguez-Echeverría S (2015) Soil changes mediated by invasive Australian acacias. Ecosistemas 24:59–66

Lorenzo P, Rodríguez J, González L, Rodríguez-Echeverría S (2016a) Changes in microhabitat, but not allelopathy, affect plant establishment after *Acacia dealbata* invasion. J Plant Ecol:rtw061. doi:10.1093/jpe/rtw061

Lorenzo P, Reboredo-Durán J, Múñoz L, González L, Freitas H, Rodríguez-Echeverría S (2016b) Inconsistency in the detection of phytotoxic effects: a test with *Acacia dealbata* extracts using two different methods. Phytochem Lett 15:190–198

Louda SM, Pemberton RW, Johnson MT, Follett PA (2003) Nontarget effects—the Achilles' heel of biological control? Retrospective analyses to reduce risk associated with biocontrol introductions. Annu Rev Entomol 48:365–396

Lowe SR, Woodford DJ, Impson DN, Day JA (2008) The impact of invasive fish and invasive riparian plants on the invertebrate fauna of the Rondegat River, Cape floristic region, South Africa. Afr J Aquat Sci 33:51–62

Luis A, Gil N, Amaral ME, Duarte AP (2012) Antioxidant activities of extract from *Acacia melanoxylon, Acacia dealbata, Olea europaea* and alkaloids estimation. Int J Pharm Sci 4:225–231

Luque GM, Bellard C, Bertelsmeier C, Bonnaud E, Genovesi P, Simberloff D, Courchamp F (2014) The 100th of the world's worst invasive alien species. Biol Invasions 16:981–985

Maoela MA, Jacobs SM, Roets F, Esler KJ (2016a) Invasion, alien control and restoration: legacy effects linked to folivorous insects and phylopathogenic fungi. Austral Eco 41:906–917

Maoela MA, Roets F, Jacobs SM, Esler KJ (2016b) Restoration of invaded Cape floristic region riparian systems leads to a recovery in foliage-active arthropod alpha- and beta-diversity. J Ins Conserv 20:85–97

Manor R, Cohen O, Saltz D (2008) Community homogenization and the invasiveness of commensal species in Mediterranean afforested landscapes. Biol Invasions 10:507–515

Marais C, Wannenburgh AM (2008) Restoration of water resources (natural capital) through the clearing of invasive alien plants from riparian areas in South Africa—costs and water benefits. S Af J Bot 74:526–537

Marchante H, Marchante E, Freitas H (2003) Invasion of the Portuguese dune ecosystems by the exotic species *Acacia longifolia* (Andrews) Willd.: effects at the community level. In: Child LE, Brock JH, Brundu G, Prach K, Pyšek P, Wade PM, Williamson M (eds) Plant invasions: ecological threats and management solutions. Backhuys Publishers, Kerkwerve, pp 75–85

Marchante H, Marchante E, Buscardo E, Maia J, Freitas H (2004) Recovery potential of dune ecosystems invaded by an exotic acacia species (*Acacia longifolia*). Weed Technol 18:1427–1433

Marchante E, Kjøller A, Struwe S, Freitas H (2008a) Short- and long-term impacts of *Acacia longifolia* invasion on the belowground processes of a Mediterranean coastal dune ecosystem. Appl Soil Ecol 40:210–217

Marchante E, Kjøller A, Struwe S, Freitas H (2008b) Soil recovery after removal of the N_2-fixing invasive *Acacia longifolia*: consequences for ecosystem restoration. Biol Invasions 11:813–823

Marchante E, Kjøller A, Struwe S, Freitas H (2008c) Invasive *Acacia longifolia* induce changes in the microbial catabolic diversity of sand dunes. Soil Biol Biochem 40:2563–2568

Marchante H, Freitas H, Hoffmann JH (2010) Seed ecology of an invasive alien species, *Acacia longifolia* (Fabaceae), in Portuguese dune ecosystems. Am J Bot 97:1780–1790

Marchante H, Freitas H, Hoffmann JH (2011a) Post-clearing recovery of coastal dunes invaded by *Acacia longifolia*: is duration of invasion relevant for management success? J Appl Ecol 48:1295–1304

Marchante H, Freitas H, Hoffmann JH (2011b) The potential role of seed banks in the recovery of dune ecosystems after removal of invasive plant species. Appl Veg Sci 14:107–119

Marchante H, Freitas H, Hoffmann JH (2011c) Assessing the suitability and safety of a well-known bud-galling wasp, *Trichilogaster acaciaelongifoliae,* for biological control of *Acacia longifolia* in Portugal. Biol Control 56:193–201

Marchante H, López-Núñez FA, Freitas H, Hoffmann JH, Impson F, Marchante E (2017) First report of the establishment of the biocontrol agent *Trichilogaster acaciaelongifoliae* for control of invasive *Acacia longifolia* in Portugal. EPPO Bulletin. doi:10.1111/epp.12373

Maslin R, McDonald MW (2004) *Acacia* search. Evaluation of *Acacia* as a woody crop option for southern Australia, RIRDC. Union Offset Printers, Canberra

May BM, Attiwill PM (2003) Nitrogen-fixation by *Acacia dealbata* and changes in soil properties 5 years after mechanical disturbance or slash-burning following timber harvest. For Ecol Manag 18:339–355

Mok HF, Majumder R, Laidlaw WS, Gregory D, Baker AJ, Arndt SK (2013) Native Australian species are effective in extracting multiple heavy metals from biosolids. Int J Phytoremediat 15:615–632

Montesinos D, Castro S, Rodríguez-Echeverría S (2012) Invasive acacias experience higher ant seed removal rates at the invasion edges. Web Ecol 12:33–37

Montesinos D, Castro S, Rodríguez-Echeverría S (2016) Two invasive acacia species secure generalist pollinators in invaded communities. Acta Oecol 74:46–55

Morais MC, Panuccio MR, Muscolo A, Freitas H (2012) Salt tolerance traits increase the invasive success of Acacia longifolia in Portuguese coastal dunes. Plant Physiol Biochem 55:60–65

Morais MC, Freitas H (2012) The acclimation potential of Acacia longifolia to water stress: implications for invasiveness. Plant Sci 196:77–84

Mostert E, Gaertner M, Holmes PM, Rebelo AG, Richardson DM (2017) Impacts of invasive alien trees on threatened lowland vegetation types in the Cape floristic region, South Africa. S Afr J Bot 108:209–222

Mukwada G, Manatsa D (2017) Acacia mearnsii management in a South African national park: SWOT analysis using hot topics in biological invasion as a guide. J Mount Sci 14:205–218

Murcia C, Aronson J, Kattan GH, Moreno-Mateos D, Dixon K, Simberloff D (2014) A critique of the 'novel ecosystem'concept. Trends Ecol Evol 29:548–553

Murphy DJ (2008) A review of the classification of Acacia (Leguminosae, Mimosoideae). Muelleria 26:10–26

Narwal SS (2010) Allelopathy in ecological sustainable organic agriculture. Allelopathy J 25:51–72

Ndou E, Ruwanza S (2016) Soil and vegetation recovery following alien tree clearing in the Eastern Cape Province of South Africa. Afr J Ecol 54:460–470

Nentwig W, Bacher S, Pyšek P, Vilà M, Kumschick S (2016) The generic impact scoring system (GISS): a standardized tool to quantify the impacts of alien species. Environ Monitor Assessment 188:1–13

Newton AC, Echeverría C, Cantarello E, Bolados G (2011) Projecting impacts of human disturbances to inform conservation planning and management in a dryland forest landscape. Biol Conserv 144:1949–1960

Olajuyigbe OO, Afolayan AJ (2012) In vitro antibacterial and time-kill assessment of crude methanolic stem bark extract of Acacia mearnsii De Wild against bacteria in shigellosis. Molecules 17:2103–2118

Panetta FD, Timmins SM (2004) Evaluating the feasibility of eradication for terrestrial weed incursions. Plant Protection Quarterly 19:5–11

Parker MA (2001) Mutualism as a constraint on invasion success for legumes and rhizobia. Divers Distrib 7:125–136

Payne SE, Kotze AC, Durmic Z, Vercoe PE (2013) Australian plants show anthelmintic activity toward equine cyathostomins in vitro. Vet Parasitol 196:153–160

Pearson DE, Callaway RM (2003) Indirect effects of host-specific biological control agents. Trends Ecol Evol 18:456–461

Pemberton RW (2000) Predictable risk to native plants in weed biological control. Oecologia 125:489–494

Perrando ER, Corder MPM (2006) Rebrota de cepas de Acacia mearnsii em diferentes idades, épocas do ano e alturas de corte. Pesqui Agropecu Bras 41:555–562

Perriot R, Breme K, Meierhenrich UJ, Carenini E, Ferrando G, Baldovini N (2010) Chemical composition of French mimosa absolute oil. J Agr Food Chem 58:1844–1849

Post JA, Kleinjan CA, Hoffmann JH, Impson FAC (2010) Biological control of Acacia cyclops in South Africa: the fundamental and realized host range of Dasineura dielsi (Diptera: Cecidomyiidae). Biol Control 53:68–75

Prinsloo FW, Scott DF (1999) Streamflow responses to the clearing of alien invasive trees from riparian zones at three sites in the Western Cape Province. Southern Afr For J 185:1–7

Rangan H, Kull CA, Alexander L (2010) Forest plantations, water availability, and regional climate change: controversies surrounding Acacia mearnsii plantations in the upper Palnis Hills, southern India. Reg Environ Chang 10:103–117

Rascher KG, Große-Stoltenberg A, Máguas C, Meira-Neto JAA, Werner C (2011a) Acacia longifolia invasion impacts vegetation structure and regeneration dynamics in open dunes and pine forests. Biol Invasions 13:1099–1113

Rascher KG, Große-Stoltenberg A, Máguas C, Werner C (2011b) Understory invasion by Acacia longifolia alters the water balance and carbon gain of a Mediterranean pine forest. Ecosystems 14:904–919

Richardson DM, Pysek P, Rejmánek M, Barbour MG, Panetta D, West CJ (2000a) Naturalization and invasion of alien plants: concepts and definitions. Divers Distrib 6:93–107

Richardson DM, Allsopp N, D'Antonio CM, Milton SJ, Rejmánek M (2000b) Plant invasions—the role of mutualisms. Biol Rev 75:65–93

Richardson DM, Kluge RL (2008) Seed banks of invasive Australian Acacia species in South Africa: role in invasiveness and options for management. Perspect Plant Ecol 10:161–177

Richardson DM, Rejmánek M (2011) Trees and shrubs as invasive alien species—a global review. Divers Distrib 17:788–809

Richardson DM, Carruthers J, Hui C, Impson FAC, Miller JT, Robertson MP et al (2011) Human-mediated introductions of Australian acacias—a global experiment in biogeography. Divers Distrib 17:771–787

Rodger JG, Johnson SD (2013) Self-pollination and inbreeding depression in Acacia dealbata: can selfing promote invasion in trees? S Afr J Bot 88:252–259

Rodríguez J, Lorenzo P, González L (2017) Different growth strategies to invade undisturbed plant communities by Acacia dealbata Link. For Ecol Manag 399:47–53

Rodríguez-Echeverría S, Crisóstomo JA, Freitas H (2007) Genetic diversity of rhizobia associated with Acacia longifolia in two stages of invasion of coastal sand dunes. Appl Environ Microb 73:5066–5070

Rodríguez-Echeverría S, Crisóstomo JA, Nabais C, Freitas H (2009) Belowground mutualists and the invasive ability of Acacia longifolia in coastal dunes of Portugal. Biol Invasions 11:651–661

Rodríguez-Echeverría S (2010) Rhizobial hitchhikers from down under: invasional meltdown in a plant–bacteria mutualism? J Biogeogr 37:1611–1622

Rodríguez-Echeverría S, Le Roux JJ, Crisóstomo JA, Ndlovu J (2011) Jack-of-all-trades and master of many? How does associated rhizobial diversity influence the colonization success of Australian Acacia species? Divers Distrib 17:946–957

Rodríguez-Echeverría S, Afonso C, Correia M, Lorenzo P, Roiloa SR (2013) The effect of soil legacy on competition and invasion by Acacia dealbata Link. Plant Ecol 214:1139–1146

Samways MJ, Caldwell PM, Osborn R (1996) Ground-living invertebrate assemblages in native, planted and invasive vegetation in South Africa. Agric Ecosyst Environ 59:19–32

Sánchez-Martín J, Beltrán-Heredia J, Rodríguez-Sánchez MT (2012) Removal of Erioglaucine (acid blue 9) with a new coagulant agent from Acacia mearnsii tannin extract. Color Technol 128:15–20

Santos M, Bastos R, Vicente J, Berger U, Soares Filho BS, Rodrigues H et al (2015) Anticipating invasions and managing impacts: a review of recent spatiotemporal modelling approaches. In: Canning-Clode J (ed) Biological invasions in changing ecosystems: vectors, ecological impacts, management and predictions. Walter de Gruyter GmbH & Co KG, Berlin, pp 389–410

Schortemeyer M, Atkin OK, McFarlane N, Evans JR (1999) The impact of elevated atmospheric CO_2 and nitrate supply on growth, biomass allocation, nitrogen partitioning and N_2 fixation of Acacia melanoxylon. Funct Plant Biol 26:737–747

Schortemeyer M, Atkin OK, McFarlane N, Evans JR (2002) N_2 fixation by Acacia species increases under elevated atmospheric CO_2. Plant Cell Environ 25:567–579

Schumann AW, Little KM, Eccles NS (1995) Suppression of seed germi- nation and early seedling growth by plantation harvest residues. S Afr J Plant Soil 12:170–172

Seastedt TR, Hobbs RJ, Suding KN (2008) Management of novel ecosys- tems: are novel approaches required? Front Ecol Environ 6:547–553

Seymour CL, Veldtman R (2010) Ecological role of control agent, and not just host-specificity, determine risks of biological control. Austral Ecol 35:704–711

Shaw R, Schaffner U, Marchante E (2016) The regulation of biological control of weeds in Europe–an evolving landscape. EPPO Bulletin 46:254–258

Simberloff D, Von Holle B (1999) Positive interactions of nonindigenous species: invasional meltdown? Biol Invasions 1:21–32

Simberloff D, Martin JL, Genovesi P, Maris V, Wardle DA, Aronson J et al (2013) Impacts of biological invasions: what's what and the way forward. Trends Ecol Evol 28:58–66

Soares PR, Duarte FT, Freitas OM, Delerue-Matos C, Figueiredo SA, Boaventura RA (2012) Evaluating the efficiency of a veg- etal coagulant in the treatment of industrial effluents. Fresenius Environ Bull 21:2413–2418

Sousa JP, Vingada JV, Loureiro S, Da Gama MM, Soares AMVM (1998) Effects of introduced exotic tree species on growth, consumption and assimilation rates of the soil detritivore *Porcellio dilatatus* (Crustacea: isopoda). Appl Soil Ecol 9:399–403

Souto XC, Bolano JC, Gonzalez L, Reigosa MJ (2001) Allelopathic effects of tree species on some soil microbial populations and her- baceous plants. Biol Plantarum 44:269–275

Souza-Alonso P, Lorenzo P, Rubido-Bará M, González L (2013) Effectiveness of management strategies in *Acacia dealbata* Link invasion, native vegetation and soil microbial community responses. For Ecol Manag 304:464–472

Souza-Alonso P, González L, Cavaleiro C (2014a) Ambient has become strained. Identification of *Acacia dealbata* Link volatiles interfering with germination and early growth of native species. J Chem Ecol 40:1051–1061

Souza-Alonso P, Novoa A, González L (2014b) Soil biochemical alter- ations and microbial community responses under *Acacia dealbata* Link invasion. Soil Biol Biochem 79:100–108

Souza-Alonso P, Guisande-Collazo A, González L (2015) Gradualism in *Acacia dealbata* Link invasion: impact on soil chemistry and micro- bial community over a chronological sequence. Soil Biol Biochem 80:315–323

Steffen W, Crutzen PJ, McNeill JR (2007) The Anthropocene: are humans now overwhelming the great forces of nature. Ambio 36: 614–621

Stone GN, Raine NE, Prescott M, Willmer PG (2003) Pollination ecology of acacias (Fabaceae, Mimosoideae). Aust Syst Bot 16:103–118

Tassin J, Rakotomanana R, Kull C (2009a) Gestion paysanne de l'invasion de *Acacia dealbata* a Madagascar. Bois et Forets des Tropiques 300:3–14

Tassin J, Médoc JM, Kull CA, Rivière JN, Balent G (2009b) Can invasion patches of *Acacia mearnsii* serve as colonizing sites for native plant species on Réunion (Mascarene archipelago)? Afr J Ecol 47:422–432

Tejada M, Gómez I, Fernández-Boy E, Díaz MJ (2014) Effects of sewage sludge/*Acacia dealbata* composts on soil biochemical and chemical properties. Comm Soil Sci Plant Anal 45:570–580

Thompson GD, Robertson MP, Webber BL, Richardson DM, Le Roux JJ, Wilson JR (2011) Predicting the subspecific identity of invasive species using distribution models: *Acacia saligna* as an example. Divers Distrib 17:1001–1014

Thompson GD, Bellstedt DU, Richardson DM, Wilson JR, Le Roux JJ (2015) A tree well travelled: global genetic structure of the invasive tree *Acacia saligna*. J Biogeogr 42:305–314

Ulm F, Hellmann C, Cruz C, Máguas C (2016) N/P imbalance as a key driver for the invasion of oligotrophic dune systems by a woody legume. Oikos 126:231–240

Van der Colff D, Dreyer LL, Valentine A, Roets F (2015) Invasive plant species may serve as a biological corridor for the invertebrate fauna of naturally isolated hosts. J Insect Conserv 19:863–875

Van Der Waal BW, Rowntree KM, Radloff SE (2012) The effect of *Acacia mearnsii* invasion and clearing on soil loss in the Kouga mountains, Eastern cape, South Africa. Land Degrad Dev 23:577–585

van Wilgen BW, Dyer C, Hoffmann JH, Ivey P, Le Maitre DC, Richardson DM et al (2011) National-scale strategic approaches for managing introduced plants: insights from Australian acacias in South Africa. Divers Distrib 17:1060–1075

van Wilgen BW, Forsyth GG, Le Maitre DC, Wannenburgh A, Kotzé JDF, van den Berg E, Henderson L (2012) An assessment of the effectiveness of a large national-scale invasive alien plant control strategy in South Africa. Biol Conserv 148:28–38

van Wilgen BW, Fill JM, Baard J, Cheney C, Forsyth AT, Kraaij T (2016) Historical costs and projected future scenarios for the management of invasive alien plants in protected areas in the Cape floristic region. Biol Conserv 200:168–177

van Wilgen BW, Wannenburgh A (2016) Co-facilitating invasive species control, water conservation and poverty relief: achievements and challenges in South Africa's Working for Water programme. Curr Opin Env Sust 19:7–17

Veldtman R, Lado TF, Botes A, Procheş Ş, Timm AE, Geertsema H, Chown SL (2011) Creating novel food webs on introduced Australian acacias: indirect effects of galling biological control agents. Divers Distrib 17:958–967

Verbrugge LN, Leuven RSEW, Van Valkenburg JLCH, Van den Born RJ (2014) Evaluating stakeholder awareness and involvement in risk prevention of aquatic invasive plant species by a national code of conduct. Aquatic Invas 9:369–381

Vicente JR, Alagador D, Guerra C, Alonso JM, Kueffer C, Vaz AS et al (2016) Cost-effective monitoring of biological invasions under global change: a model-based framework. J Appl Ecol 53:1317–1329

Viljoen BD, Stoltsz CW (2008) Control of black wattle (*Acacia mearnsii* De Wild.) seedlings with Garlon herbicide applied by backpack mistblower: short communication. S Afr J Plant Soil 25:242–244

Werner C, Zumkier U, Beyschlag W, Máguas C (2010) High competi- tiveness of a resource demanding invasive acacia under low resource supply. Plant Ecol 206:83–96

Wilson JR, Gairifo C, Gibson MR, Arianoutsou M, Bakar BB, Baret S et al (2011) Risk assessment, eradication, and biological control: global efforts to limit Australian acacia invasions. Divers Distrib 17:1030–1046

Wimberger K, Nowak K, Hill, RA (2017). Reliance on exotic plants by two groups of threatened samango monkeys, *Cercopithecus albogularis labiatus*, at their southern range limit. Int J Primatol 38:151–171

Wintola OA, Otang WM, Afolayan AJ (2017) The prevalence and per- ceived efficacy of medicinal plants used for stomach ailments in the Amathole District municipality, Eastern Cape, South Africa. S Afr J Bot 108:144–148

Wood AR, Morris MJ (2007) Impact of the gall-forming rust fungus *Uromycladium tepperianum* on the invasive tree *Acacia saligna* in South Africa: 15 years of monitoring. Biol Control 41:68–77

Yañez R, Romaní A, Garrote G, Alonso JL, Parajó JC (2009) Experimental evaluation of alkaline treatment as a method for en- hancing the enzymatic digestibility of autohydrolysed *Acacia dealbata*. J Chem Technol Biot 84:1070–1077

Yañez R, Gómez B, Martínez M, Gullón B, Alonso JL (2013) Valorization of an invasive woody species, *Acacia dealbata*, by means of ionic liquid pretreatment and enzymatic hydrolysis. J Chem Technol Biot 89:1337–1343

Yelenik SG, Stock WD, Richardson DM (2004) Ecosystem level impacts of invasive *Acacia saligna* in the South African Fynbos. Restor Ecol 12:44–51

Zhou L, Bi Y, Jiang L, Wang Z, Chen W (2012) Effect of black wattle (*Acacia mearnsii*) extract on blue-green algal bloom control and plankton structure optimization: a field mesocosm experiment. Water Environ Res 84:2133–2142