




Article

# Pine Stand Density Influences the Regeneration of *Acacia saligna* Labill. H.L.Wendl. and Native Woody Species in a Mediterranean Coastal Pine Plantation

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**Abstract:** Mediterranean plantations are the most suitable areas to assess vegetation dynamics and competitive interactions between native and exotic woody species. Our research was carried out in a coastal pine plantation (Sicily) where renaturalization by native species (*Pistacia lentiscus* L. and *Olea europaea* var. *sylvestris*) and invasion by *Acacia saligna* (Labill.) H.L.Wendl. simultaneously occur. The regeneration pattern of woody species in the pine understory was evaluated in six experimental plots along a stand density gradient, from 200 to approximately 700 pines per hectare. Both pine stand density and regeneration by native species had a significant negative relationship with *Acacia* natural regeneration. *Olea* regeneration was positively correlated with stand density, while *Pistacia* showed a non-significant relationship. Saplings of both native species were mostly less than 1 m high, whereas approximately 70% of *Acacia* individuals were higher than 1 m. We found that 400 pines per hectare should be considered a minimum stand density to keep *Acacia* under control, while favouring the establishment of native species in the understory. The successful control of *Acacia* requires an integrated management strategy, including different forest interventions according to stand density: thinning, control measures against *Acacia*, and renaturalization actions.

**Keywords:** invasive alien plants; afforestation; renaturalization; silviculture; *Pinus pinea* L.; *Pinus halepensis* Miller; wood; forest management

## 1. Introduction

The massive afforestation activities carried out on a large scale in the Mediterranean basin during the 20th century made extensive use of alien tree species [1,2]. The need to ensure rapid soil cover and the highly degraded conditions of substrates and slopes over large areas determined a clear preference for trees belonging to the genera *Acacia*, *Eucalyptus*, and, predominantly, *Pinus* [3], due to their fast-growing, pioneer traits, ecological plasticity, and easy propagation [4]. Another important reason was the supposed role of preparatory species: Introduced species were considered ideal for improving site conditions to allow recovery of native woody species. However, in many cases, afforestation/reforestation projects have slowed down, hindered, or completely stopped the natural evolutionary dynamics and the maturation of more complex, diversified, and stable Mediterranean

forest ecosystems [5,6]. The role played by forest plantations has been a matter of intense debate in the Mediterranean [7,8], due to the marked heterogeneity of the ecological conditions and management practices. Furthermore, owing to prevailing purposes of soil protection, autoecological aspects have not been adequately considered in the choice of plant species and propagation material, thus determining the unsuitability of many afforested sites. Hence, in most cases, forest interventions are necessary to allow the gradual replacement of alien planted species by native ones. However, such necessary interventions (e.g., thinning) have not been performed or they have been performed without the guidance of clear forest planning and management objectives. In some cases, this has led to the complete failure of plantations, which have neither adequately protected the soil, enhanced local succession dynamics, nor provided adequate biodiversity conservation. As a result, the current main management choice for Mediterranean plantations is renaturalization, i.e., the gradual conversion towards more stable, complex, and diversified ecosystems, dominated by native broadleaved and evergreen trees; the manifestation of potential climax vegetation [9,10]. This objective is an absolute priority for protected coastal habitats, which have undergone rarefaction and marked alteration because of intense human exploitation of the territory.

Massive reforestation activity has provided a huge, experimental forest laboratory where we can observe and evaluate the evolution and dynamics of woody vegetation in the Mediterranean, as influenced by competition and/or facilitation interactions between native and exotic woody species [8,9]. New combinations between different tree species with varied characteristics, such as life-history strategies, origin, reproductive, and physiological traits, have resulted in artificial ecosystems, characterized by novel competitive interactions whose outcomes are not easily predictable nor can they be considered to have reached steady state [11]. They are strongly influenced by both local environmental and management conditions, as well as by the type of species that come into contact. In some cases, introduced species have shown a remarkable ability to adapt to the new ecosystem, achieving performance levels greater than that of the native co-existing species, with abundant regeneration abilities and quick soil coverage, thus, displaying invasive behaviour [12]. Such evidence further complicates the management of afforested areas in which interventions carried out to promote renaturalization may, at the same time, trigger the invasion process. This so-called paradox of ecological restoration occurs when the disturbance required to restore a forest ecosystem, which has been highly simplified by human intervention, may, on the contrary, favour invasive spread, bringing about significant negative ecological impacts [13]. Generally, invasive alien species are particularly able to exploit increases in available resources (light, nutrients) found in gaps or areas with reduced forest cover, such as those resulting from forest utilizations or restoration treatments, or even through wildfires and grazing [13,14]. Hence, selective cuttings are performed to counteract alien tree species of the overstory, while favouring the progressive replacement by native woody species, starting from the underlying layers [3].

Thinning within plantations is strictly necessary. On the one hand, such periodic interventions allow the enhancement of available resources for remaining trees, including light, soil water, and nutrients. Individual trees may grow better and increase in diameter size, and they may be healthier and less susceptible to pathogens, disease, and abiotic factors, thus rendering the whole forest stand more resistant and resilient. On the other hand, thinning has been proved to ameliorate understory microclimatic conditions, especially in terms of light available for native seedling establishment, growth, and development in coastal pine forests [15]. However, invasive tree species may also benefit from interventions, greatly hindering the natural dynamics and recovery of the functionality of forest ecosystems. Therefore, for the aforementioned reasons, active forest management, which includes appropriate silvicultural interventions, is mandatory.

Due to their high invasive potential many *Acacia* species are considered as ecosystem engineers [12], causing serious and long-lasting ecological impacts on entire ecosystems [16], including below-ground soil microbial communities [17] and native pollinators [18], as well as on soil and vegetation characteristics [16]. Some general traits seem to have favoured the worldwide

distribution of Acacias as invasive species: Nitrogen symbiotic fixation, rapid initial growth rates, large seed production, and reproductive potential. The introduction of a new functional trait or of a new biochemical process into a native ecosystem is bound, generally, to bring about greater and longer-lasting ecological consequences on ecosystems [19]. The more diverse an invasive plant species is, compared to native communities, the greater the likelihood of it having larger negative impacts. Even after eradication, the altered conditions caused by an invasive species may remain for a longer period of time [20]. Emblematic cases are the invasion of Mediterranean coastal ecosystems by *Carpobrotus* spp. [21] and by Acacias [17,22].

*Acacia saligna* (Labill.) H.L.Wendl. (hereinafter, *A. saligna*) is one of the most widely used tree species in coastal-areas afforestation, covering approximately 600,000 hectares worldwide of which almost 10% is in South Africa alone [23]. It has been preferentially used in Mediterranean-climate areas to bind dunes and stabilize soils, along road systems, and to recover bare and degraded lands, in addition to use as a windbreak [24]. Whilst functioning reasonably well in soil protection, a number of peculiar traits make *A. saligna* one of the invasive species most suited to Mediterranean coastal ecosystems, even with limited nutrient availability. Traits include high growth rates, early achievement of sexual maturity, striking resprouting ability following cutting or other physical disturbance factors, and drought and abiotic stress tolerance, as well as its ability to establish symbiosis for nitrogen fixation [25,26]. *A. saligna* may have a profound effect on the structure and functioning of recipient ecosystems and seriously hamper vegetation dynamics of woody native species, significantly reducing their richness and cover [27]. *A. saligna* may cause a significant shift in native community assemblages, reducing the occurrence of guiding or focal species, whilst favouring the spread of opportunistic and ruderal species, including other invasive species [28,29]. In Mediterranean pine plantations, negative effects caused by *A. saligna* have been reported on small mammal communities and were attributed to the structural simplification and homogenization of the forest stand [30].

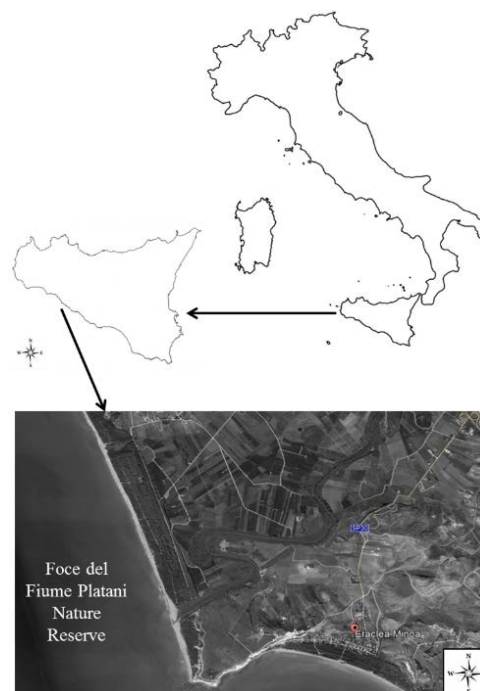
Only recently, in Italy, *A. saligna* has been listed amongst alien species posing a threat to native species and to almost all habitats of community interest linked to sandy shores and dune systems, including wooded dunes with *Pinus pinea* L. and/or *Pinus pinaster* Aiton (Habitat 2270) [29,31]. *A. saligna* was introduced to Sicily in the late nineteenth century [32], whilst the first cases of naturalization date back to the early years of the last century [33]. *A. saligna* has increasingly spread in the last decades, mainly invading the coastal dune habitats of SW Sicily and seriously threatening the vegetation communities dominated by species of considerable scientific and biogeographic interest, such as *Retama raetam* subsp. *gussonei* (Webb) Greuter [34] and *Juniperus oxycedrus* subsp. *macrocarpa* (Sm.) Ball [35]. The same plant communities are the most exposed to invasion by *A. saligna* in other Italian regions [31], as well as in other Mediterranean-type ecosystems [36]. Hence, urgent interventions of control and eradication have been invoked [37,38].

We assessed the regeneration of *A. saligna* and of co-occurring native woody species in the understory of a Mediterranean coastal plantation along a gradient of pine stand density, ranging from 200 to 700 stems per hectare. In our study site, two natural processes of opposing significance occur simultaneously: On the one hand, active processes of renaturalization of native woody species, particularly *Olea europaea* L. var. *sylvestris* (Mill.) Lehr. and *Pistacia lentiscus* L. (hereinafter, *Olea* and *Pistacia*) and, on the other hand, invasion by an alien tree species (*A. saligna*). Whilst the most favorable pine density values for the establishment and recruitment of Mediterranean native woody species are known, at least in part [39], only a limited body of knowledge exists on density values that allow the renaturalization process to occur without triggering plant invasion. Yet, this is crucial information for ecosystems where native and alien species coexist in the understory and compete. We hypothesized that there could be a threshold density value which favours the renaturalization process, whilst preventing invasive spread. Overall, we provide the most recommended management options for forest pine plantations, including specific forest interventions addressing the pine canopy (thinning), *A. saligna* (eradication), and native woody species (renaturalization).

## 2. Material and Methods

### 2.1. Study Area and Vegetation

Field surveys were carried out within the nature reserve “Foce del fiume Platani”, a 3.5 km long coastal belt consisting of 207 hectares, localized in the Agrigento Province (SW Sicily, Figure 1). Average annual precipitation is 496.7 mm, while the average annual temperature is 18.7 °C (Osservatorio delle Acque, 2016). Hence, the study area falls within the Thermomediterranean upper dry bioclimatic belt [40]. The dominant soil types in the study area are Typic Xeropsamments entisols, with a sandy texture in the first meter of soil depth [41]. The reserve includes the alluvial plain originated by the delta of the Platani River. This coastal strip has not been affected by marked anthropic alterations and still hosts residual aspects of coastal dune and Mediterranean maquis vegetation, with high potential biodiversity, linked to the presence of the sea, coastal dunes, and backdune habitats. The woody vegetation is characterized by the coexistence of Mediterranean native species, such as *Ephedra fragilis* Desf., *Pistacia lentiscus* L., *Chamaerops humilis* L., and *Olea europaea* var. *sylvestris*, with planted alien tree species, especially *Pinus* spp., *Acacia saligna* and *Eucalyptus camaldulensis* Dehnh., and alien shrub species like *Myoporum tenuifolium* G.Forster [41]. The study area also harbours plant species of particular conservation value, such as *Retama raetam* subsp. *gussonei* and *Juniperus turbinata* Guss., characteristic of the association *Ephedro-Juniperetum macrocarpae* Bartolo, Brullo & Marcenò 1982 [34].



**Figure 1.** The geographic location of the “Foce del Fiume Platani” Nature Reserve on the southwestern coast of Sicily.

### 2.2. Forest Management and Disturbance History

The forest plantation in the study area was established in 1952 [41], with the main aim of stabilizing sand dunes and backdunes, and as a windbreak to protect agricultural areas inland [24]. The plantation was mostly made with exotic tree species: Pines (*Pinus pinea*, *Pinus halepensis* Mill. and *Pinus canariensis* C.Sm.), eucalypts (*Eucalyptus camaldulensis* and *Eucalyptus occidentalis* Endl.), and Acacias, such as *Acacia cyclops* G.Don and *Vachellia karroo* (Hayne) Banfi & Galasso. *A. saligna* was also planted, although, interestingly, its felling began just 15 years later to favour the establishment of

the pine forest [42]. However, until 1975, forest interventions were mainly limited to the elimination of dead trees. Subsequently, thinning was periodically carried out with the elimination of one tree row every three rows. The actual nature reserve was then established in 1984, with the decree 216 of the Regional Department of Territory and Environment, and, in 1988, it was assigned to the Regional Department of Rural and Territorial Development (DRSRT). The main goals were, “to ensure the conservation of bird communities, to promote the re-establishment of Mediterranean maquis vegetation, of halophilic plant communities and of dune fauna” (free authors’ translation from the original decree). Over the last two decades, periodic understory clearing and thinning of the exotic species were carried out to promote the gradual conversion towards native-species-dominated forest ecosystems. However, the increasing spread of *A. saligna* in the understory is making the achievement of this desirable goal extremely difficult. Continuous selective thinning interventions were always performed without the guidance of a forest management plan, whilst the control of *A. saligna* was never clearly addressed. No wildfires or grazing have occurred in the protected site over the last 50 years.

### 2.3. Experimental Plots and Sampling Design

To assess the natural regeneration performed by *A. saligna*, field surveys were carried out in 2017 over six experimental plots along a density gradient of dominant pine (*Pinus pinea* and *Pinus halepensis*), ranging from 200 to about 700 stems per hectare. Such values fall among the most commonly observed in thinned Mediterranean pine afforestations [43–45]. *Pinus* spp. are the only tree species composing the upper layer. Each study plot consisted of a 21 × 21 m square (441 m<sup>2</sup>) and was established after an in-depth investigation of all the study area to ensure homogeneity of main environmental factors (e.g., soil characteristics, slope, stoniness, herbaceous cover, distance from the sea) and silvicultural history. Specifically, the altitudinal difference was <5 m, the distance between plots was <500 m, all plots were located in a range of 250–350 m from the sea, and they were established with a minimum distance of 20 m from unpaved roads to minimize potential edge effects. All target tree species were present in all plots and in their surroundings. Hence, we considered the pine stand density as the most relevant driving factor of tree regeneration in the understory.

Within each plot, besides identification of the *Pinus* species, the following dendrometric parameters were surveyed for each individual tree: Diameter at breast height (DBH =  $D_{1.30\text{ m}}$ ), total tree height, and basal area. To assess natural regeneration by native woody species and by *A. saligna*, three 5 × 5 m subplots (25 m<sup>2</sup>) were established within each plot. Natural regeneration was evaluated in terms of species identity, density (N seedlings or saplings m<sup>-2</sup>), origin (seed-borne or vegetative sprout), and development (diameter at root collar, height of the largest saplings). We considered only regenerating individuals higher than 5 cm and the following four height (H) classes:

- Class 1:  $H \leq 50$  cm;
- Class 2:  $50\text{ cm} < H \leq 100$  cm;
- Class 3:  $100\text{ cm} < H \leq 200$  cm;
- Class 4:  $200\text{ cm} < H \leq 400$  cm.

Within the upper height classes (3 and 4), we measured the diameter at the root collar in 10 randomly chosen individuals from each class. To determinate the age and the growth rate of *A. saligna*, we collected three discs from the stem of mature individuals above ground level (diameters = 3.0, 8.5 and 9.5 cm). The surface of the stem discs was sanded with progressively finer grades of sandpaper (up to 1000 grits) to produce a flat, polished surface on which tree ring boundaries were clearly visible under magnification. Growth rings were counted under the microscope (Leica Microsystems, Heerbrugg, Switzerland, Leica DFC 420C©) along three radii per disc, with close attention paid to the occurrence of wedging, missing, and/or false rings.



## 2.4. Statistical Analysis

Before performing statistical analysis, the assumption of data normality was assessed through the Anderson-Darling test. Both *A. saligna* ( $p > 0.15$ ) and the native species ( $p > 0.15$ ) regeneration density, as well as that of the pine stand ( $p > 0.15$ ), followed a normal distribution. Linear regression analysis ( $p < 0.05$ ) was used to assess the correlation between the density of *A. saligna* and the pine stand, and between the density of *A. saligna* and the native woody species. We also assessed differences between the height classes, regardless of woody species, and within each woody species between the height classes. As *A. saligna* data achieved normality after log transformation (Anderson-Darling Test;  $p = 0.114$ ), we performed a one-way ANOVA followed by a Tukey's HSD (Honestly Significant Difference) test. As *Olea* data achieved normality after log transformation (Anderson-Darling Test;  $p = 0.127$ ), but variance was unequal, we performed a one-way ANOVA followed by a Tamhane's T2 Test. As *Pistacia* data did not achieve normality, we performed a Kruskal-Wallis test (non-parametric) followed by a Conover-Inman Test. As data within each height class were not normally distributed, we again performed a Kruskal-Wallis test (non-parametric) followed by a Conover-Inman Test. Statistical analysis was performed using Systat Software, Inc., San Jose, CA, USA, 2009 (Version No.13.00.05).

## 3. Results

### 3.1. Dominant Pine Cover

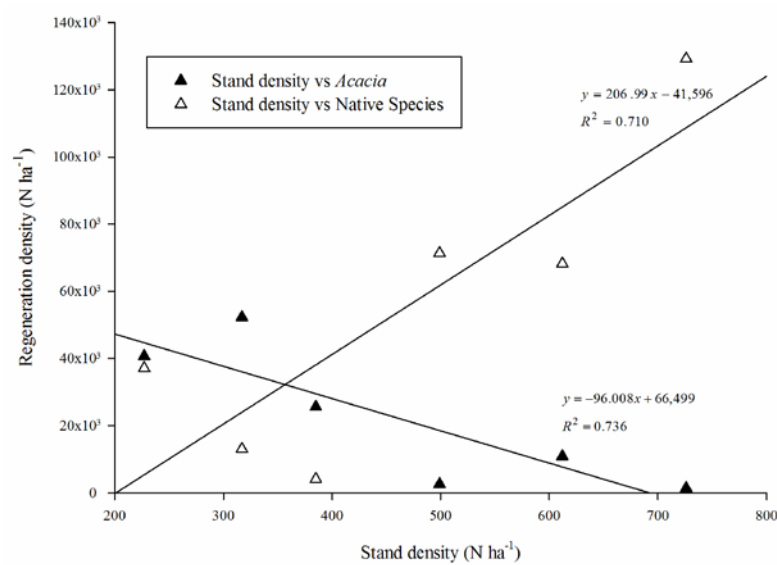
Density and dendrometric parameters (DBH, height, and basal area) of the pine canopy cover showed a clear pattern (Table 1). First, a strong positive ( $r^2 = 0.99$ ) correlation between tree density and tree size, in terms of basal area, was found.

**Table 1.** Main dendrometric parameters of the pine canopy layer of the study plots. Means are followed by standard deviations.

Plot	Dominant Tree Species	Density (N ha <sup>-1</sup> )	DBH (cm)	Height (m)	Basal Area (m <sup>2</sup> ha <sup>-1</sup> )
1	<i>Pinus halepensis</i>	227	17.3 ± 6.6	5.5 ± 1.3	9.7
2	<i>Pinus halepensis</i>	317	21.3 ± 5.6	6.8 ± 0.4	12.0
3	<i>Pinus halepensis</i>	385	23.6 ± 7.0	6.8 ± 1.3	18.2
4	<i>Pinus pinea</i>	499	24.2 ± 1.8	9.1 ± 0.6	23.0
5	<i>Pinus pinea</i>	612	23.3 ± 3.7	10.3 ± 0.5	29.7
6	<i>Pinus pinea</i>	726	23.4 ± 3.9	9.2 ± 0.3	34.1

### 3.2. Regeneration Layer

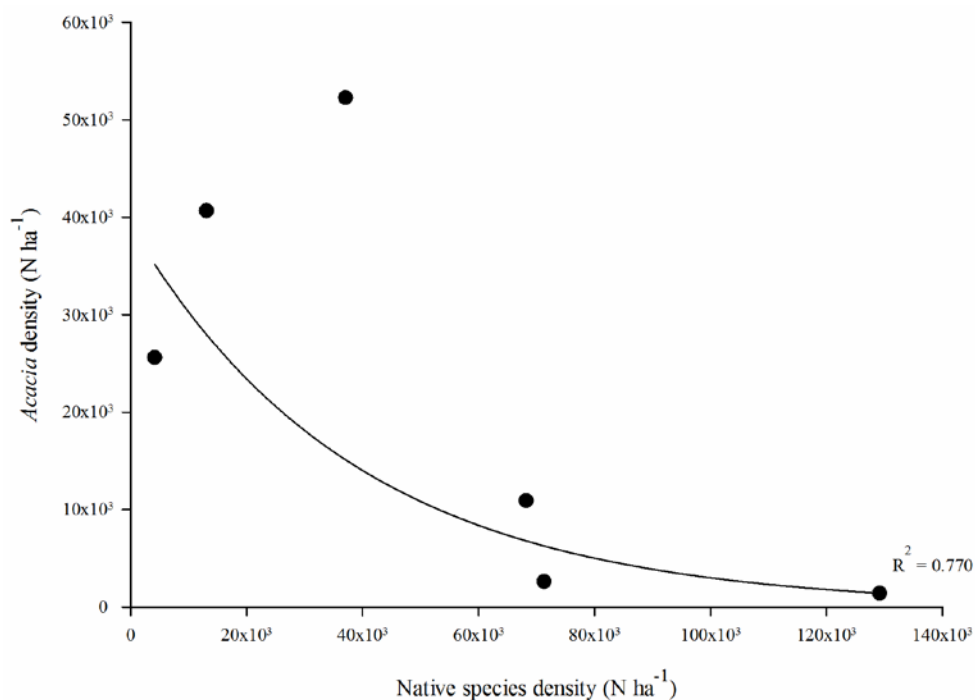
A non-significant positive correlation was found between the overall regeneration density (native species plus *A. saligna*) and the pine stand density ( $F = 2.80$ ,  $p = 0.169$ ,  $r^2 = 0.412$ ). When analysed separately (Figure 2), *A. saligna* regeneration was negatively correlated ( $F = 11.15$ ,  $p = 0.029$ ,  $r^2 = 0.736$ ), whereas the native woody species regeneration was positively correlated to the pine density ( $F = 9.79$ ,  $p = 0.035$ ,  $r^2 = 0.710$ ). In effect, only *Olea* regeneration was positively correlated ( $F = 9.82$ ,  $p = 0.035$ ,  $r^2 = 0.711$ ) to the pine density, showing *Pistacia* a non-significant relationship ( $F = 0.31$ ,  $p = 0.607$ ,  $r^2 = 0.072$ ).



**Figure 2.** Linear regression analysis between the pine stand density and the native woody species and *Acacia saligna* regeneration.

The intersection between the two trend lines allowed us to find a reference value, corresponding to a pine stand density of  $356.75 \text{ ha}^{-1}$ .

A strong negative exponential relationship ( $r^2 = 0.770$ ) was found between the regeneration density of the native woody species and *A. saligna* (Figure 3).



**Figure 3.** Relationship between the density of the native woody species and *Acacia saligna* regeneration.

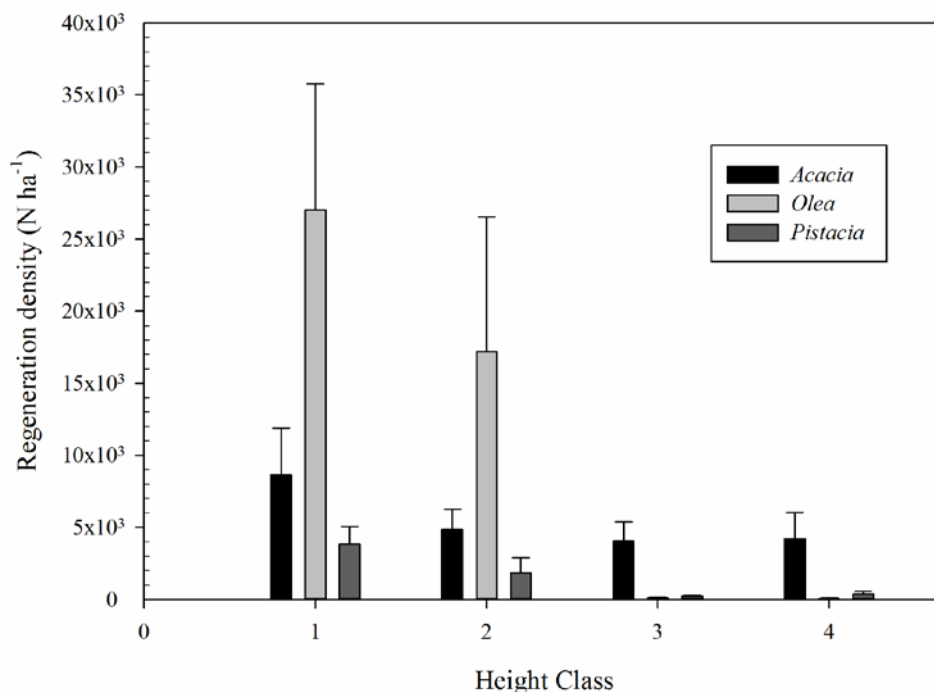
Overall, regeneration was found to be mainly concentrated in the smallest height classes (1 and 2), together accounting for more than 85% of all the individuals (Table 2). Height classes 3 and 4 accounted for a little over 12% of all the individuals, which was almost exclusively represented by *A. saligna*. *Olea* reached the highest regeneration density (mean value: 4.7 individuals  $\text{m}^{-2}$ ), followed by *A. saligna* (mean value: 2.2 individuals  $\text{m}^{-2}$ ), and *Pistacia* (mean value: 0.6 individuals  $\text{m}^{-2}$ ).

However, large differences, in terms of height class, were found between species. *Olea* regeneration was clearly dominant in the smallest height classes (1 and 2), whilst *A. saligna* was by far the most prevalent in height classes 3 and 4. In terms of abundance, height classes 1 and 2 were not different from each other, but both were significantly higher than height classes 3 and 4, which, in turn, were equal with each other (Kruskal-Wallis 23.815,  $p < 0.001$ ). Regarding *A. saligna* regeneration, we found very little variation in the diameter at the root collar in relation to plant height, with values ranging from 1 to 1.5 cm in height class 3 and from 2 to 3 cm in height class 4.

**Table 2.** Distribution in height class (%) of the woody species regeneration. The relative contribution (%) of each species to each height class is reported within round brackets. Means are followed by the standard error.

Woody Species	Height Class (%)			
	1	2	3	4
<i>Acacia saligna</i>	39.8 ± 8.9 (21.9)	22.3 ± 3.5 (20.3)	18.6 ± 3.7 (92.3)	19.3 ± 11.6 (90.9)
<i>Olea europaea</i>	60.9 ± 15.9 (68.4)	38.7 ± 15.9 (72.0)	0.2 ± 0.1 (2.4)	0.2 ± 0.1 (1.4)
<i>Pistacia lentiscus</i>	61.3 ± 15.6 (9.7)	29.3 ± 17.2 (7.7)	3.7 ± 3.1 (5.3)	5.7 ± 2.3 (7.8)
Total	54.5	33.0	6.1	6.4

No significant differences between the height classes were found in *A. saligna* Labill. regenerating individuals ( $F = 0.975$ ,  $p = 0.414$ ) (Figure 4). In *Olea*, height classes 1 and 2 were not different from each other and both were significantly higher than height classes 3 and 4 which, in turn, were equal with each other ( $F = 16.498$ ,  $p < 0.001$ ) (Figure 4). In *Pistacia*, height class 1 was significantly higher than the other height classes. Height class 2 was significantly higher than height classes 3 and 4 which, in turn, were equal with each other (Kruskal-Wallis 22.971,  $p < 0.001$ ).



**Figure 4.** Height class distribution of the woody species regeneration in the understory. Vertical bars show the standard error of the mean ( $\pm$ SE).

Regarding wood samples, it must be underlined that there are divergent results from different authors about the presence of growth rings in *A. saligna*. According to InsideWood database [46] and



El-Sahhar et al. [47], the growth rings in this species are indistinct, whereas Crivellaro [48] showed that the growth rings boundaries are distinct. In our wood samples, the growth rings of *A. saligna* were visible in the wood anatomy as bands of marginal parenchyma that run around the entire disc. Counts of parenchyma bands produced disc ages of 6–12 years. We found that the average growth rate was 0.7 cm per year; similar results were found in Kenya by Jama [49].

#### 4. Discussion

*Acacia saligna* is one of the most widespread invasive species in the Mediterranean basin, where it has been widely used for afforestation purposes and it now threatens native biodiversity and alters ecosystem structure and functioning of large areas. On the other hand, in many areas, Mediterranean afforestations are significantly affected by renaturalization processes by native woody species. Both processes are increasingly common in the Mediterranean basin and they require appropriate management practices. In the understory of a Mediterranean pine plantation, the regeneration pattern of woody native species and the invasive *A. saligna* were found to be significantly affected by pine stand density and, also, by forest management. The overall management of pine afforestation within the Platani Reserve would require varying types of forest interventions: Thinning of the dominant pine canopy, control measures against *A. saligna*, and renaturalization actions to favour native woody species.

##### 4.1. Pine Management and Thinning

Stand density and dominant tree size both contribute to determining the amount of available resources in the understory, thus, affecting the competitive dynamics between woody species and altering the complex balance of facilitation and inhibition interactions [5,7,8,50]. Light, as well as soil water and nutrient content are known to play a crucial role in pine understory dynamics [5,43,51]. This is particularly true in Mediterranean coastal pine forests where water supply during summer is considered the main limiting factor of plant growth [9]. Very high stand densities (approximately 1150 pines per hectare) have strong inhibitory effects for shrub species, resulting from a reduction in light, soil water, and nutrient availability [6]. There is a large consensus and much field evidence regarding the negative role played by an excessive pine cover in the renaturalization process of Mediterranean pine plantations, as well as in the richness of the understory [8,13,30]. For this reason, thinning is, generally, performed with the aim of accelerating secondary succession and increasing the overall biodiversity, enhancing the heterogeneity and structural diversity of the forest stand and improving the status of the remaining trees [7,13,52]. Different thinning intensities are usually required for different forest types and species [52,53].

Within Mediterranean pine afforestations, moderate thinning is considered the most suitable option to enable the establishment of most native woody species, including *Olea* and *Pistacia*, with a recommended density ranging from approximately 330 to 550 pines per hectare [39,54,55]. Densities greater than 500 pines per hectare are generally required for thermophilous oaks, such as *Quercus ilex* L., which are shade-demanding at the seedling stage [8,56,57]. However, previous research was carried out in the absence of light-demanding invasive species, such as *A. saligna*. The variation in pine stand density in our study site depends on periodic thinning that felled, mainly, the largest tree individuals, while the remaining trees have not yet had enough time to grow. Indeed, larger pines were found in areas with higher stand density. Conversely, the number of individuals per unit area and average tree size are inversely correlated in natural forest stands [58]. Our research suggests the need to adopt different forest management practices according to the pine stand density. With densities of up to 400 pines per hectare, no thinning should be performed as a further decrease in stand density would cause the undisputed spread of *A. saligna*. A density greater than 400 is the minimum threshold to contain the invasive potential of *A. saligna*, while favouring the recruitment and establishment of native species. Densities higher than 500 pines per hectare were related to an increase in the regeneration density of approximately 6.5-fold for *Olea* and 2-fold for *Pistacia*, together with a concomitant decrease

in *A. saligna* of approximately 8-fold. Low-intensity, localized, and gradual thinning is strongly recommended for densities exceeding 600 pines per hectare in which higher shading has kept *A. saligna* relatively under control. Mediterranean woody species, such as *Quercus* spp. and *Olea*, which may survive under a closed canopy during the early stages of life, need more light in subsequent sapling and juvenile phases [57]. Such necessary interventions should be preceded by the clearing of any adult *A. saligna* plants to avoid the risk of a new invasion.

#### 4.2. Regeneration Pattern in the Understory

In our study site, we assessed the current regeneration pattern of *Acacia saligna* and native woody species in the understory of a Mediterranean coastal pine plantation. We found evidence of a species-specific response, with *A. saligna* being negatively correlated to the stand density, *Olea* being positively correlated, and *Pistacia* being not significantly correlated. At high pine densities (>500 pines per hectare), *Olea* regeneration is by far the most prevalent, while *A. saligna* regeneration is prevalent under low pine densities (<400 pines per hectare). At intermediate conditions (400–500 pines per hectare), *Pistacia* tends to prevail slightly, while both *A. saligna* and *Olea* are represented by just over 21% of regenerating individuals. As only current regeneration has been assessed, it should be considered that long-term fluctuations could arise and cannot be excluded.

*Olea* and *Pistacia* showed quite different behaviour based on the stand density. *Olea* attained an impressive regeneration density that linearly increased with stand density, with the maximum value exceeding 120,000 seedlings per hectare (mean: 47,110 N ha<sup>-1</sup>). Previous regeneration data obtained from field investigations carried out in the same protected area suggest that *Olea* has experienced a marked increase in the last 10–15 years [41]. Such a sizable change may be due to an increase in the effective seed dispersal by birds and to possible shifts in their natural population [59]. Conversely, *Pistacia* showed a much lower regeneration density (mean: 5,700 N ha<sup>-1</sup>), with non-significant variations based on the stand density. Such a pattern was not completely unexpected as *Pistacia* was found to dominate the understory in high pine cover conditions [30] and its considerable ecological plasticity to solar radiation is recognized [60]. However, saplings of both native species were much less developed than *A. saligna*, being mostly less than one meter high in *Olea* (>99%) and *Pistacia* (>91%). Conversely, approximately 70% of *A. saligna* saplings were taller than one meter, suggesting a clearly higher reproductive potential.

#### 4.3. Acacia Management

Our field surveys and widespread evidence collected worldwide suggest that the control and eradication of *A. saligna* is a difficult and time-consuming task. After heavy thinning or a natural disturbance event, this species seems to be better equipped than native species to occupy free space and exploit available resources, thus proving more competitive. Under such circumstances, *A. saligna* may exert strong negative effects on the growth and establishment of native woody species, seriously hindering the renaturalization process and any chance of an autonomous evolution towards more complex and diversified forest stands. *A. saligna* has proven to be particularly favoured by high light availability, such as can be found in natural gaps or recently cut areas [29]. However, its regeneration has also been found in high pine densities, underlining its remarkable adaptability to a wide spectrum of light conditions [50]. Furthermore, pine cover may have played a key role in the initial stages of invasion, offering suitable microclimatic conditions for regeneration in the backdunes and providing protection from direct sun exposure, marine salt-spray, and frequent winds, as well as, presumably, higher soil water and nutrient availability [45,51].

The temporal dynamics of *A. saligna* invasion should be considered carefully as time has significant consequences for the efficacy of control actions. The ecological impacts *A. saligna* exerts on the recipient ecosystem [16] and the average time needed for ecosystem recovery may vary significantly with the duration of the invasion. The alterations in soil chemical characteristics, such as increased pH and soil organic matter and nutrients (N and P), are time-dependent effects that are generally associated

with *A. saligna* [27,61,62]. For fast-growing trees, like Acacias, a time span of 20–25 years is already considered a long-term invasion [61,62]. A factor explaining the massive regeneration of *A. saligna* in recent decades could be the improvement in edaphic conditions due to *Acacia* litter accumulation, as was found for *Acacia cyclops* on the Island of Lampedusa [63]. Time has also allowed *A. saligna* to express its maximum reproductive potential. Annually, *A. saligna* mother plants may release approximately 5,400 seeds per square meter to the soil [64]. While less than 10% of seeds germinate in the first year, the remaining seeds help to build up a consistent and long-lasting soil seed bank [65,66]. In South African invaded stands, more than 44,000 seeds per square meter may be found in the seed bank, including the soil and litter layers [67,68]. This huge amount depends on the long viability of the seed in the soil, resisting up to 50 years in the absence of mechanical scarification [65] and even 10 years after clearing [62,66,68]. Resistance to fire events is also highly influential for Mediterranean ecosystems [65]. *A. saligna* could have reached its maximum potential soil seed bank in the Platani nature reserve as this is related to the stem diameter reaching 6 cm [68], a value commonly found in our field surveys.

The successful management of *A. saligna* can be achieved only if its reproductive potential is destroyed or at least severely limited. The interventions required should aim to reduce the population of adult individuals and, therefore, to curb the current seed production and to deplete seed banks by destroying seeds and/or triggering mass germination [65]. Regarding clearing treatments, excluding the use of herbicides, a possible option could be felling and burning for the combined effect on *A. saligna* living trees and regeneration, as well as on the soil seed bank [27,64,66]. However, in our protected site, the use of prescribed burning is not recommended due to difficulties in controlling the spread of fire outside well-confined areas and, also, due to negative ecological effects on the overall biodiversity. Simply stopping seed fall may lead to reductions in the soil seed bank of 80% in four to six years [67]. Hence, it is crucial to intervene with systematic cuttings of *A. saligna* mature plants before dissemination, i.e., every year before summer. This is not simple as *A. saligna* reaches reproductive maturity extremely quickly: We observed individuals at just one meter tall already fruiting in the study area. It is also essential to eliminate seedlings that may emerge after a disturbance event [68]. The absence of continuous forest interventions may lead to the comeback of the seed bank at pre-intervention levels in only seven years [67]. After stand clearing, the altered soil conditions caused by the invasion of *A. saligna* can last up to 10 years [62] and may, especially, threaten the presence of native species which have a narrow ecological niche and are accustomed to ecosystems with limited nutrient availability [69]. In our study site, it is highly likely that no fewer than 10 years of continuous and capillary interventions will be necessary for the effective eradication of *A. saligna*. Considering this, an effective possibility could be the production of woodchips of *A. saligna* and other wood residuals (i.e., from pruning, thinning, ecc.).

Another factor that further complicates the management of pine afforestation is the concomitant occurrence of other non-native tree species, such as *Acacia cyclops* A.Cunn. ex G.Don and *Pinus canariensis* [70], *Eucalyptus occidentalis* Endl. [71], *Vachellia karroo* (Hayne) Banfi & Galasso, and *Myoporum tenuifolium* G.Forst. All these species experienced full naturalization in the protected site and could spread further in the near future, completely altering local vegetation dynamics, with unpredictable consequences. Such potentially invasive species may benefit from the increased nutrient availability provided by *A. saligna* and the clearing treatments against it. The threat also posed by potential secondary invasion has to be carefully evaluated during the necessary periodic monitoring of the study area [72].

#### 4.4. Renaturalization

After eradication, especially in areas with sufficient cover of native woody species, specific interventions addressing renaturalization of the forest plantation could be carried out. It is necessary to assess whether local species can autonomously recover their role in the ecosystem, starting from the understory. This opportunity depends on various ecological, management,

and landscape characteristics. Three main aspects must be considered: Soil seed bank dynamics, the dispersal ability of native species, and the occurrence and abundance of suitable seed dispersers. Firstly, it should be noted that *A. saligna* is able to reduce the soil seed bank of native species over time [61]. Forest ecosystems in the surrounding areas of the Platani reserve are very rare and fragmented; mature woody individuals, therefore, are too few and/or too far away, thus compromising the chance of passive restoration processes via bird communities [73]. Active restoration through direct sowing or planting of native woody species is required [8,27,62]. The effective renaturalization is, however, quite difficult to achieve as many biotic and abiotic factors limiting seed and seedling establishment and affecting mortality rates need to be considered. For instance, Li et al. [74] found that the successful strategy to reintroduce native woody species in the understory of conifer plantations is using seedlings older than six months or fencing the area.

Then, as native woody species in the Platani reserve all bear fleshy fruits or acorns, their dissemination strongly depends upon the occurrence of seed dispersers, especially frugivorous birds [73,75,76]. Gómez-Aparicio et al. [8] found that 4 km is the maximum seed dispersal distance by *Garrulus glandarius* L. to enable the effective establishment of Holm oak in Mediterranean contexts. Some scattered regenerating individuals of *Quercus ilex* and *Quercus pubescens* Willd. s.l. have been found in the study area, probably due to dissemination by the jay [76]. The recovery of woody vegetation could be relatively fast as many bird seed dispersers, such as the Eurasian jay (*Garrulus glandarius* Linnaeus), the song thrush (*Turdus philomelos* Brehm), the common starling (*Sturnus vulgaris* Linnaeus), and the blackbird (*Turdus merula* Linnaeus) are already present in the protected site or in its surrounding area (T. La Mantia and R. Bueno, pers. obs.).

## 5. Conclusions

In Mediterranean pine plantations threatened by *A. saligna*, forest interventions should differ according to pine stand density. The present case study may offer many useful pointers for the management of similar Mediterranean afforestations where thinning, aimed at favouring renaturalization by native woody species, could, otherwise, trigger the spread of invasive alien species, thus counterbalancing the positive effects of necessary interventions. In the absence of any active management in the Platani reserve, an inexorable increase in the invasion process by *A. saligna* is likely. Adaptive forest management capable of maintaining sufficient pine cover in the areas most at risk of invasion, while increasing the light available in areas with excessive stand density, appears to be the best trade-off to foster natural succession processes, whilst preventing invasion by *A. saligna*. The complex understory dynamics and competitive interactions between native species and *A. saligna* highlight the need for periodic interventions, as well as for follow-up treatments and regular and constant monitoring.

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