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Regeneration of potential laurel forest under a native canopy and an exotic canopy, Tenerife (Canary Islands)

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Abstract

Exotic tree monocultures adversely affect native ecosystems through competition and through alteration of nutrient availability and dynamics. However, evidence suggests that some tree plantations facilitate a more rapid recovery of native plant communities by providing shelter to the plants below and by attracting seed dispersers. The results are specific to the plantation species and the type of native forest. In this study, we analyzed the differences in regeneration of native woody species in two stands of exotic species, *Pinus radiata* and *Eucalyptus globulus*, occupying native laurel forest ground. We also examined the process of regeneration in two adjacent native forest stands to determine whether the exotic plants negatively affect the recovery of the native plant community.

The native stands differed from the exotic stands in tree species richness and basal area. However, regeneration was similar in alien and native stands. Differences were quantitatively important, but the species composition of the regenerating community was similar. Moreover, these planted areas recovered their canopy quickly because *P. radiata* and *E. globulus* have rapid growth in comparison with native laurel forest species. The rapid recovery of the canopy has prevented more serious erosion damage and has to some extent facilitated the regeneration of native species. The results of this study indicate that a progressive elimination of the exotic stands would favor the establishment and growth of native species. The role of some exotic tree species in the restoration of laurel forest areas should not always be seen as negative.

Key words: exotic plantations; foster effect; laurisilva; canopy basal area; seedlings; DCA; ecological restoration.

Resumen

Regeneración en zonas potenciales de laurisilva bajo una cubierta nativa y una cubierta de exóticas, Tenerife (Islas Canarias)

Los monocultivos de especies exóticas afectan a los ecosistemas nativos a través de la competencia con especies nativas y alteración de la disponibilidad de nutrientes. Sin embargo, existen evidencias de que las plantaciones de exóticas pueden facilitar la recuperación de las comunidades de plantas nativas proveyendo protección para los renovales y atrayendo los dispersores de semillas. En cualquier caso los resultados son específicos de cada plantación y de la especie exótica utilizada. En este estudio se analizan las diferencias de regeneración de especies leñosas en dos formaciones de especies exóticas, *Pinus radiata* y *Eucalyptus globulus*, en zonas potenciales de laurisilva y también se examina la regeneración en dos bosques adyacentes de laurisilva y se comparan las diferencias de las plantaciones de exóticas en la regeneración.

La formación de laurisilva se diferencia de las de exóticas en riqueza y área basimétrica. Sin embargo, la composición de la regeneración es muy similar entre las formaciones de nativa y exóticas. Las diferencias son importantes cuantitativamente pero la composición de especies de la comunidad de plantas regenerando es similar. Las plantaciones han recuperado la cubierta muy rápidamente ya que *P. radiata* y *E. globulus* son especies de crecimiento rápido en comparación con las nativas. Esta recuperación rápida de la bóveda ha prevenido los daños erosivos y ha facilitado la regeneración de las especies nativas.

Los resultados indican que una progresiva eliminación de la formación de exóticas favorecería el establecimiento y crecimiento de las especies nativas. Las especies exóticas pueden jugar un papel importante en la restauración del bosque de laurel y no siempre tienen que asumirse como elementos negativos.

Palabras clave: plantaciones exóticas; efecto refugio; laurisilva; área basimétrica; DCA; restauración ecológica.

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Introduction

Because of conflicting evidence, it is not yet known whether exotic monocultures exert a predominantly negative or positive effect on the regeneration of the native forest beneath their canopies. Many exotic tree monocultures adversely affect native ecosystems through competition and alteration of nutrient cycles (Attiwill and Leeper, 1987; Jurgensen *et al.*, 1986; Fimbel and Fimbel, 1996). However, evidence suggests that some tree plantations facilitate a more rapid restoration of the plant forest community by providing shelter to the plants below and by attracting seed dispersers (Arévalo and Fernández-Palacios, 2005; Arévalo *et al.,* 2005; Powers *et al.*, 1997). The use of nurse crops to aid the establishment of desirable species has been a common practice in northern Europe. The nurse-crop approach, understood as a natural successional sequence from the initial herb and shrub stage to the climax stage, resembles a process in which pioneer tree species are gradually replaced by climax species (Pommerening and Murphy, 2004). Primarily, coniferous species have been used as nurse trees. This practice requires subsequent transformation of these even-aged forests into uneven-aged continuous forest (Schütz, 2001). In either case, the results are specific to the plantation species and the native forest type (Haggar *et al.*, 1997; Parotta 1995; Parotta *et al.*, 1997).

In this context, plantations are receiving new attention because of the positive effect they seem to have on the regeneration of the native forest beneath their canopy. The «catalytic», «nurse crop» or «foster ecosystem» effect of plantations on the succession and recovery of the native forest appears to be a widespread phenomenon and has not yet been analyzed sufficiently (Harris and Harris, 1997; Lugo, 1997; Parotta, 1993; Parotta, 1995; Parotta *et al.*, 1997; Whitmore, 1999; Arévalo and Fernández-Palacios 2005). The direct catalytic effects of plantations include accelerating the rate of succession compared with the normal early successional series (Lugo, 1992; Geldenhuys, 1997; Powers *et al.*, 1997), kickstarting the regeneration of later successional species earlier than the immature status of the earlier series would normally allow (Ashton *et al.,* 1997; Franklin *et al.,* 1999), and overcoming arrested successions in which the natural succession is held back by some degrading influence, such as the loss of soil productivity (Haggar *et al.*, 1997).

The catalytic effect of plantations is not thought to alter the direction of succession from the previous vegetation type when used in areas that can potentially support forests. This principle seems to apply equally well to plantations of exotics and plantations of native species. The effect of a plantation on native forest recovery has also been interpreted as positive in terms of the socioeconomic value of the plantation products (Chaubey *et al.*, 1988; Whitmore, 1999) and in terms of the diversity it adds to the landscape (Murcia, 1997).

Both *Pinus radiata* and *Eucalyptus globulus* plantations have received mixed reviews. The divergent effects of *Eucalyptus* on various ecosystems of the world have sparked «the *Eucalyptus* controversy» (Loumeto and Huttel, 1997). Although work on plantations is now an «international research effort» (Parotta *et al.*, 1997), almost no work has been done in the Canaries, and the few studies that have been conducted in other cloud forest show a negative impact on native forest regeneration (Murcia, 1997).

Afforestation on the island of Tenerife has been performed mainly with an endemic pine species (*Pinus canariensis*), the dominant species of the native pine forest stands. However, the exotic *Pinus radiata* D. Don and *Eucalyptus globulus* Labill. ssp. *globulus* have been introduced in some areas. These areas include our study area, Agua García, which is enclosed in the protected area of «Paisaje Protegido de Las Lagunetas.» The most apparent human disturbance in Agua García, a potential area of laurel forest, is the existence of exotic plantations. The effect of *Pinus radiata* and *Eucalyptus globulus* plantations on the recovery of the laurel forest is unknown. The entire forest of Agua García, except for one small patch, was cleared approximately 50 years ago (Brito and Lucía, 1985). One-fifth of the cleared area has been planted with pine or eucalyptus as part of a nationwide effort to increase exports. The desirability of the plantations is now in question in light of the recognized importance of the native laurel forest. Currently, however, no specific administrative plans exist for promoting the laurel forest, but restoration of natural vegetation and the elimination of exotic vegetation have been promoted by local authorities during the past fifteen years (Machado, 2001) for the eradication of different species as *Pinus radiata*, *Eucalyptus globulus* or *Nicotiana glauca* (García *et al.*, 1995).

In this study, we aim to evaluate the effect of plantations with exotic species on the regeneration of native woody species of laurel forest in Canary Islands, following the general hypothesis of a catalytic effect of the canopy created by exotic planted species on the recovery of the native forest. In particular, this study aims to test the following hypotheses: i) exotic species have a positive effect on the regeneration of the laurel forest vegetation in the stands studied, and ii) exotic species regenerate under their own canopy. The results of the study may suggest some management approaches that can assist those responsible for these protected areas in selecting the best management practices for the conservation and the restoration of potential laurel forest.

Material and methods

Study sites

The study was conducted in two potential laurel forest areas (Ceballos and Ortuño 1974) in the NE of Tenerife: Agua García (28° 27' N, 16° 24' W) and Anaga (28° 32' N, 16° 17' W) (Fig. 1).

The study site in Agua García is characterized by different vegetation types (forest, fields, open forest, and others), and it is located in the «Paisaje Protegido de Las Lagunetas». The area is used regularly by locals and tourists for picnicking, horse, bicycle and motorcycle riding, rabbit hunting and pine litter collection (Brito and Lucía, 1985). Of the ca. 80 ha of the park, approximately 30 ha have been converted to *Pinus radiata* plantations and 10 ha to *Eucalyptus* plantations. The afforestation was conducted in the 1940s. In the areas where reforestation programs were not conducted, the potential vegetation has recovered naturally (Aboal 1998) while not in others. Subsequently, the plantations have had no further treatments. The densities of the stands vary between 900-2,900 trees/ha. The bedrock of Agua García is a mixture of 4-5 My old volcanic ash and olivinic basalt with high permeability and drainage (Morales *et al*., 1996). The soil is ca. 1 m deep stony, colluvial andosol, divided relatively evenly

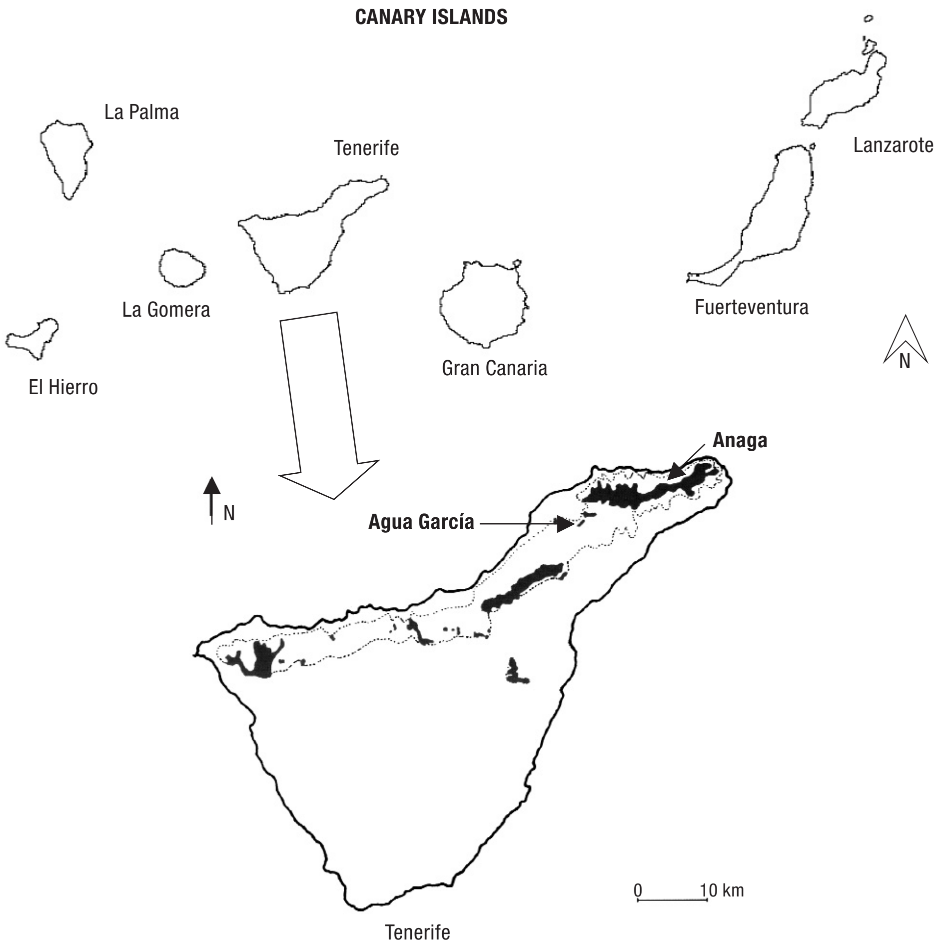

Figure 1. Map of the Canary Islands and Tenerife island, indicating the location of the study as well as the potential laurel forest (surrounded by the dotted line) and the remaining laurel forest (in dark).

between a reddish clay A-horizon and a more andic, loamy B-horizon. The mean annual temperature is 13.6°C. Mean temperatures range between 10.5°C in February and 16.6°C in July (García Gallo and Wildpret, 1990). Frost does not occur. Annual precipitation reaches 756 mm. The area has the summer droughts characteristic of the Mediterranean climate. Nevertheless, the cloud layer formed by the trade winds results in high relative humidity (annual median of 82.1%), and it can double the effective amount of precipitation (Kämmer, 1974).

The canopy can reach 24 m in the *Eucalyptus* and *Pinus* plantations. The understory is relatively poor in species and is dominated by *Erica arborea*, *E. scoparia* and *Chamaecytisus proliferus*. Further information about species composition in the area can be found in Arévalo and Fernández-Palacios 2005.

The other area selected for study, Anaga Rural Park, is ca. 12 km to the east of Agua García, in the NE corner of Tenerife. The park encompasses a 7- to 8-million-year-old basaltic massif (Ancochea *et al.,* 1990) covering some 130 km^2. The park represents 7% of Tenerife's total area. The soils have been classified in the order Entisol, suborder Orthens, and they are typical of high slope areas. They maintain a high humidity due to the hydrated aluminium silicates that they contain, and they have a thick A horizon (Fernández-Caldas *et al.*, 1985). Organic matter content is high (~10%), and pH is approximately 5.5 (Fernández-Palacios and Arévalo, 1998). The mean annual temperature is close to 15°C and has minimal annual and daily fluctuations. Frost does not occur. Two seasons, winter and summer, can be distinguished, but in most years, differences between the two most extreme months are not great (differences between the averages of the extreme months: 8°C, 5% relative humidity and 100 mm of rain) (Ceballos and Ortuño, 1974). The annual precipitation in the park reaches 900 mm, but it can be twice this amount because of fog drip.

The canopy height of Anaga's laurel forest (the potential vegetation of this area and of Agua García) can reach 10-20 m, depending on the slope. Maximum heights are found at basin bottoms. Canopy height decreases progressively toward the basins' borders. The laurel forest of Anaga contains a total of 19 tree species (Santos, 1990). Dominant species include *Laurus azorica*, *Erica scoparia*, *Erica arborea*, *Ilex canariensis*, *Prunus lusitanica*, *Myrica faya* and *Viburnum tinus*. The dominance of a given species depends on site conditions. For example, *E. scoparia* dominates in forest ridges, *L. azorica* in mesic zones and *E. arborea* in more disturbed areas (Anon., 1973). Further information on stand composition, structure and environment in the study sites can be found in Arévalo and Fernández-Palacios (1998), Arévalo *et al.* (1999), and Arévalo and Fernández-Palacios (2000).

The relict Canarian laurel forest has been exploited and transformed since the 15th century, following European colonization (Parsons, 1991). The two study areas differ greatly in conservation quality. Agua García is a mosaic of different woody vegetation types and vehicle trails, and it has a complex history of disturbance and exploitation. The Anaga forests are far better preserved. Following the reduction and fragmentation of the primary area, Agua García experienced sequential clearcutting of the original vegetation, planting of exotic timber species, removal of these and, finally, regrowth of the native laurel forest under several silvicultural regimes.

Anaga, in contrast, includes the largest, oldest and least disturbed laurel forest on Tenerife. The degree of wood harvesting is lower in Anaga than in Agua García. This lower level of disturbance by harvesting contributes to the greater integrity of the forest. In both forests the main dividing elements are paved and unpaved roads and paths, and impacts include car traffic and human access to relatively well-preserved areas. Thus, Agua García is a good area to test the effect of plantations on the regeneration of natural stands. Based on the above information, we chose to treat Anaga as the reference area and Agua García as the disturbed area (except for a control site in Agua García).

Experimental design

To evaluate the effect of plantations on laurel forest regeneration, we selected six stands: a 12-15 m tall *Eucalyptus globulus* stand (dominant height; plots 1-8), a 30-35 m tall *Pinus radiata* stand (plots 9-16), two well-conserved forest stands at Anaga and Agua García (plots 17 to 20 and plots 21 to 24, respectively) and two managed laurel forest stands in Anaga and Agua García (plots 25 to 28 and plots 29 to 32, respectively). The management measures used in the two last-named stands consisted in the cutting of the sprouts of the trees (leaving only one dominant stem per coppice) every 7-8 years for agricultural purposes. These practices were abandoned approximately 15-20 years ago.

The general quadrat size used in the study was 25 × 25 m. In the managed laurel forest, with lower canopies and dense and prickly vegetation, we used 12 × 12 m plots, as suggested by Kent and Coker (1992). In each plot, we measured the diameter at breast height (DBH) for all individual trees greater than 50 cm in height and larger than 2.5 cm DBH. From this information, we obtained the basal area of the plots and their tree species richness (number of species). We also counted all individuals smaller than 50 cm height (seedlings) in four 2 × 2 m subplots at each corner of the plot to determine the identity of the species that were regenerated and established under the different canopies, and we calculated the density and richness for these smaller individuals. The dominant height per plot was also measured. Sampling was carried out during June and July 2000.

Statistical analysis

Ordination techniques help to explain community variation (Gauch, 1982), and they can be used to evaluate trends over time and space (Franklin *et al.*, 1999; Arévalo *et al.*, 1999; Ter Braak and Šmilauer, 1998). We used Detrended Correspondence Analysis (DCA; Hill and Gauch, 1980) using the CANOCO package of Ter Braak and Šmilauer (1998) to examine how species composition changed through space and to determine whether patterns based on different forest stands could be extrapolated from the analyses. In the plane given by DCA axes I and II, we encircled the plots of the same stand with an envelope, using the minimum possible area of the space. We performed the analysis separately for the basal area of canopy trees and for seedling densities in the 32 plots.

Differences in basal area, tree species richness, seedling species richness and seedling density were analyzed using the Kruskal-Wallis test ($p < 0.05$). If significant differences were found, we used post hoc nonparametric Tukey tests. Basic statistical methods followed Zar (1984) and were implemented using the SPSS statistical package (Anon., 1986).

Results

All the plots of the four classified stands were in the potential area of laurel forest in Tenerife, with a northerly dominant aspect and an altitude between 700-1,100 m (Table 1).

Analysis of the basal area data from the sites (Table 2 and Fig. 2a) shows that basal area was significantly larger in the *Pinus radiata* stand (47.08 m^2/ha; $\chi^2 = 10.92$, k = 4, $p < 0.01$), followed by the laurel forest stand (34.97 m^2/ha) and the *Eucalyptus* and managed laurel forest (20 m^2/ha, approximately, for both). The density values of the plots were highly variable. No trends related to regeneration could be extracted from this variable (Table 2).

Eighteen canopy tree species were widely distributed over the plots (Table 2, Fig. 2b). Of these trees, 9 were laurel forest species. As expected, *Eucalyptus globulus* and *Pinus radiata* plantations had either monospecific or very species-poor canopies. Some individuals of *Erica arborea, Myrica faya* and *Laurus azorica* were present. *L. azorica* can reach heights of 15-20 m in light gaps. In contrast, laurel forest reference plots had significantly higher values of canopy species diversity (6; $\chi^2 = 14.07$, k = 4, $p < 0.05$). This value of canopy richness is almost double that of the other stands.

We found significantly higher seedling densities in control laurel forest and in managed laurel forest than in the other stands (Fig. 2c; 80 seedlings/m^2, $\chi^2 = 14.07$, k = 4, $p < 0.01$). Seedlings of nine tree species (*Laurus azorica, Erica arborea, Myrica faya, Ilex canariensis, Viburnum tinus, Pinus radiata, Picconia excelsa, Ocotea foetens and Apollonias barbujana*) were found at the

Table 1. Environmental variables of the different stands (average values)

	Slope sex (°)	Altitude (m)	Aspect	Cover (%) Rock	Cover (%) Soil	Cover (%) Litter	Shrub cover (%)	Canopy cover (%)
Eucalyptus stand	5	1,050	N	5	10	80	60	70
Radiata stand	5	1,070	N	5	5	95	65	65
LF* Agua García	10	830	N	5	6	55	25	90
LF forest Anaga	20	750	NW	8	5	60	15	95
MLF** Agua García	5	980	NW	8	5	60	90	85
MLF forest Anaga	15	800	N	10	10	60	30	85

*Laurel Forest. **Managed Laurel Forest.

Table 2. Canopy species richness, total density (trees/ha), mean diameter (m) and basal area (m²/ha) of canopy species in the different sampled plots

Plot	*Erica arborea*	*Eucalyptus globulus*	*Ilex canariensis*	*Ilex perado*	*Laurus azorica*	*Myrica faya*	*Persea indica*	*Pinus radiata*	Tree sps. richness	Total density	Mean diam.	Basal area
1	—	12.80	—	—	—	—	—	—	1	393	0.10	12.80
2	—	66.92	—	—	—	—	—	—	1	152	0.37	66.92
3	—	46.06	—	—	—	—	—	—	1	286	0.23	46.06
4	—	6.69	0.04	—	0.03	0.45	—	—	4	2,100	0.03	7.21
5	0.05	4.56	—	—	—	—	—	—	2	1,339	0.03	4.61
6	0.05	9.32	—	—	—	—	—	—	2	1,617	0.04	9.37
7	0.57	15.00	—	—	0.08	—	0.65	—	4	1,982	0.05	16.30
8	0.39	13.55	—	—	0.22	—	—	—	3	1,036	0.07	14.16
9	—	—	—	—	—	0.04	—	50.48	2	1,792	0.09	50.52
10	2.65	—	—	—	0.57	1.16	0.07	42.50	5	1,200	0.11	46.95
11	3.33	—	—	—	—	—	—	33.03	2	1,528	0.09	36.36
12	—	—	—	—	0.20	—	—	53.67	2	1,692	0.10	53.87
13	—	—	—	—	0.13	—	—	60.43	2	1,740	0.11	60.56
14	—	—	—	—	0.54	—	—	41.66	2	2,898	0.07	42.20
15	0.29	—	—	—	6.01	—	—	34.53	3	2,664	0.07	40.83
16	—	—	—	—	0.28	—	—	59.59	2	948	0.14	59.87
17	9.40	—	1.45	—	11.41	9.01	—	—	4	956	0.10	31.27
18	5.59	—	—	—	3.95	10.95	4.87	—	4	353	0.15	25.36
19	9.44	—	0.54	0.90	9.29	13.49	2.07	—	9	122	0.31	35.73
20	10.50	—	8.86	0.43	9.66	1.17	—	—	10	577	0.16	43.71
21	0.38	—	0.11	20.02	30.22	8.60	1.09	—	9	1,659	0.11	61.51
22	0.50	—	4.98	1.62	24.97	4.60	1.00	—	6	1,511	0.10	43.05
23	1.50	—	—	—	15.54	5.44	—	—	3	1,422	0.07	22.48
24	0.40	—	0.44	—	6.06	9.78	—	—	6	680	0.09	16.68
25	2.27	—	1.59	—	1.90	11.39	—	—	4	1,914	0.05	17.15
26	1.27	—	—	—	0.23	10.97	—	—	3	628	0.08	12.47
27	3.18	—	0.79	—	1.92	6.08	—	—	4	2,329	0.04	11.97
28	12.48	—	6.87	—	8.16	3.10	—	—	4	3,415	0.05	30.61
29	7.21	—	2.08	—	3.71	48.57	—	—	4	1,920	0.10	61.57
30	4.09	—	—	—	3.72	4.77	—	—	3	1,002	0.06	12.58
31	1.23	—	—	—	7.08	2.36	—	—	3	2,825	0.03	10.67
32	5.46	—	—	—	—	1.06	—	—	2	1,824	0.03	6.52

Also: *Prunus lusitanica* (12.95 m²/ha), *Erica scoparia* (0.14) in 24; *Picconia excelsa* (0.96), *Heberdenia excelsa* (0.08), *Viburnum tinus* (0.05) in 19 and *Heberdenia excelsa* (3.71), *Apollonias barbujana* (0.98), *Rhamnus glandulosa* (0.11), *Viburnum tinus* (0.47), *Visnea mocanera* (0.11) in 20.

study sites (Table 3 and Fig. 2d). All of these species occurred in the laurel forest stand, whereas 3-5 of the species occurred in the other stands. Species richness was also significantly higher in the laurel forest (2.8 species/100 m²; $\chi^2 = 10.97$, $k = 5$, $p < 0.05$). As stated elsewhere (Fernández-Palacios and Arévalo, 1998; Arévalo and Fernández-Palacios, 2003), *L.azorica* is the dominant seedling species in the laurel forest. The number of seedlings increased significantly in these managed stands, relative to other stands. This increase results directly from the management measures that remove some of the adult individuals and thereby favor the establishment of seedlings.

The DCA analysis of the basal area data for trees revealed a strong principal DCA axis that clearly discriminated the stands of *Pinus radiata* and *Eucalyptus globulus* plantations from the stands of laurel forest and managed laurel forest. The degree of variability was much lower in the *E. globulus* and *P. radiata* stands, as indicated by the small polygon that enclosed the six plots of each stand. A secondary gradient along the DCA represented variation inside the laurel forest, much more heterogeneous than the plantation stands. The homogeneous composition of the plantation treatments aligned them around a very small area in the bidimensional DCA space (Fig. 3).

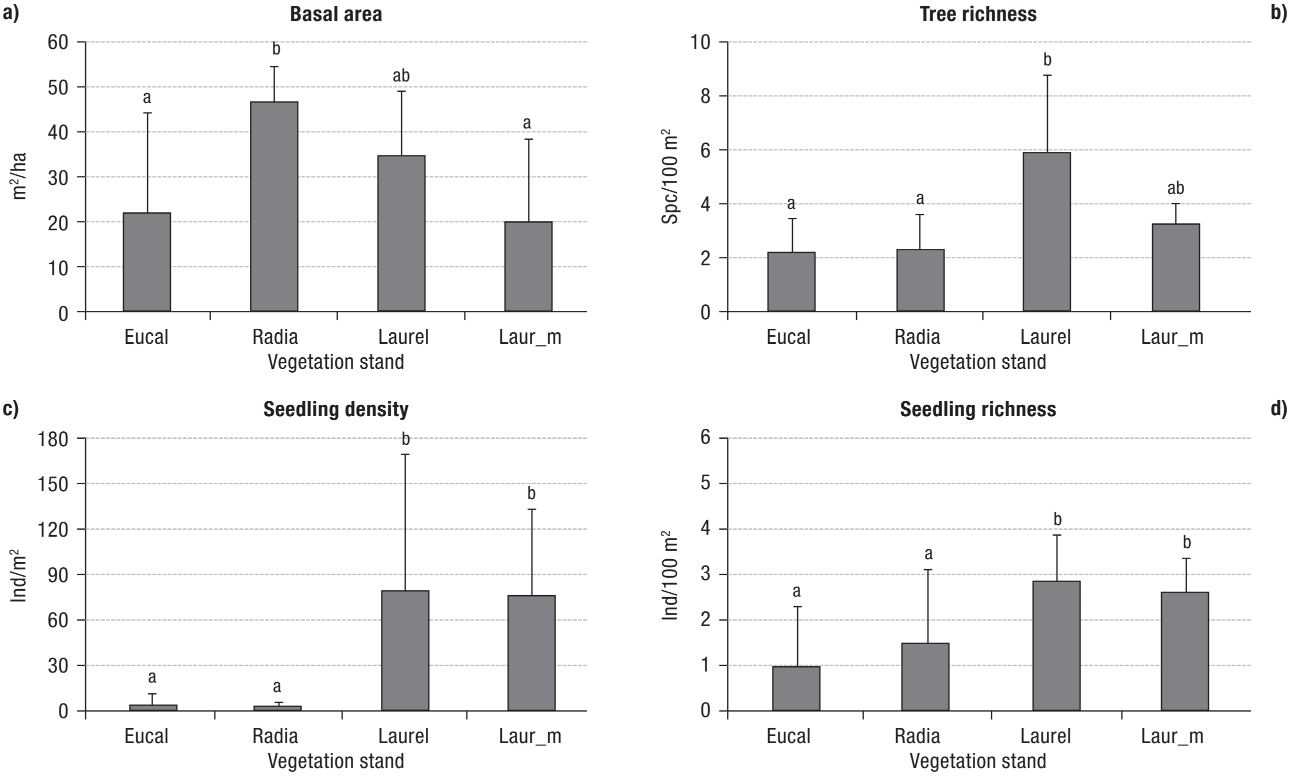

Figure 2. Mean and standard deviation of the a) basal area of each forest type, b) tree species richness, c) seedling density and d) seedling richness of each forest stand. Identical letters above the bars indicate non-significant differences indicated by the non parametric post-hoc Tukey test (for $p < 0.05$) (Eucal for the Eucalyptus globulus stand, Radia for the Pinus radiata stand, Laurel for the laurel forest stand and Laur_m for the managed laurel forest stand).

The DCA of seedling density (Fig. 4) produced a different result. The different stands analyzed exhibited considerable overlap. The center of the seedling data in DCA space corresponded to the location of the most abundant seedling species, *Laurus azorica*. The ordination did not discriminate any of the stands. All stands except for the eucalyptus stand were enclosed within the envelope defined by the *Pinus radiata* stands. This result reflects the convergence of the regeneration processes among sites. The characteristics found for regeneration were independent of the differences found in the canopy structure and composition of the sites. Species having low seedling densities were located along the borders of the graph.

Discussion

Eucalyptus plantations are possibly the most widespread and controversial of all plantation types (Florence, 2004). The results of this study suggest that this type of plantation produces poorer regeneration, compared with the laurel forest stands. These plantations also show the smallest basal area of laurel forest trees and contain virtually no seedlings (Table 2). The few stems observed under the eucalyptus canopy belonged to *Erica arborea*, a pioneer wind-dispersed species. However, at a low density of *E. arborea*, germination should be stimulated by light (Arévalo and Fernández-Palacios, 1998).

In contrast, *Pinus radiata* plantations are known to support diverse vegetation beneath their canopy (Fimbel and Fimbel, 1996) and can even produce the «foster effect» (Lugo, 1992; Arévalo and Fernández-Palacios, 2003). In this study, the pine plantations demonstrated significantly lower laurel forest species basal area and richness than the control (but higher than the *Eucalyptus* stands).

The seedling species richness of these exotic stands is low in contrast with the native laurel forest or even the managed laurel forest. However, the seedling diversity is much greater than the diversity of canopy trees. This result demonstrates a colonization effect that can be applied to the preliminary establishment of

Table 3. Regeneration composition (seedlings/m^2) of the different sampled plots

Plot number	*Erica arborea*	*Ilex canariensis*	*Laurus azorica*	*Myrica faya*	*Viburnum tinum*	Seedling number/m^2	Seedling sp. richness
1	—	—	—	—	—	0	0
2	—	—	—	—	—	0	0
3	—	—	—	—	—	0	0
4	1	—	22	1	—	24	3
5	1	1	1	—	—	3	3
6	—	—	2	—	—	2	1
7	1	—	—	—	—	1	1
8	—	—	—	—	—	0	0
9	—	—	—	—	—	0	0
10	1	—	8	—	—	9	2
11	—	—	8	—	—	8	1
12	—	—	—	—	—	1	1
13	—	—	3	—	—	4	2
14	—	2	1	—	—	3	2
15	—	—	8	—	10	18	2
16	—	—	1	—	1	2	2
17	—	4	63	—	1	68	3
18	—	1	49	—	2	52	3
19	—	22	32	—	—	54	2
20	—	—	19	—	2	21	2
21	—	1	17	—	3	41	5
22	—	—	7	—	5	13	3
23	—	—	269	10	10	289	3
24	—	—	97	—	3	100	2
25	—	—	157	—	7	164	2
26	—	—	19	—	5	24	2
27	—	3	49	—	7	59	3
28	—	—	59	—	10	69	2
29	—	—	89	—	10	99	2
30	—	—	134	1	14	149	3
31	—	1	6	—	10	17	3
32	2	1	22	—	4	29	4

Also: *Pinus radiata* (1) in 12 and 13; *Picconia excelsa* (19) and *Apollonias barbujana* (1) in 19; and *Ocotea foetens* (1) in 20.

native forest if the exotic species are removed. The differences found among stands are quantitatively important, but qualitatively (*i.e.,* species composition), regeneration composition is similar across the stands and cannot be discriminated among the stands (Fig. 4). Moreover, the planted areas have recovered their canopy very rapidly. They would do so as long as the growth of *Pinus radiata* and *Eucalyptus globulus* was rapid in comparison with the growth of native laurel forest species (Arévalo, 1998). This rapid recovery of the canopy has prevented more profound erosion damage and has facilitated the regeneration of native species to some degree.

Until now, *Pinus radiata* and *Eucalyptus globulus* have not become invasive in Tenerife. In other areas, they have been found to be invasive, but this invasive character has also been found to be relatively uncommon (Rejmanek *et al.,* 2005). In this study, we found no regeneration of *P. radiata* and *E. globulus*, probably due to the more ecologically pioneer character of these species with respect to the species of the laurel forest. The extent of the area in which they occur is the same as it was when they were planted approximately 40-50 years ago. These results suggest that conservation-restoration efforts have not to be devoted to invasion control but to removal of current *E. globulus* plantations and gradual thinning of *P. radiata* if the final objective is to convert the current plantations to original forest in a relatively short-time. Thinning of the *P. radiata* stand will need to involve a balance between the arresting effect of the stand and its paradoxical potential for producing a «nurse crop» of the few *Laurus azorica*

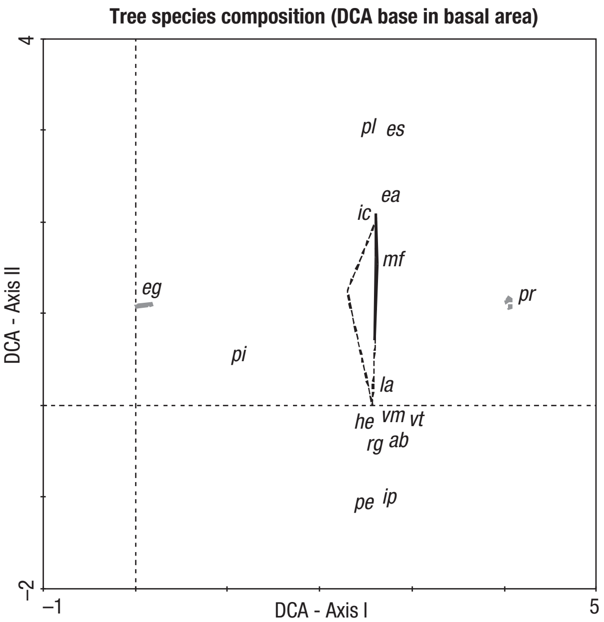

Figure 3. Species scores in the ordination space defined by axis I and axis II of the DCA derived from basal area. Polygons enclose the plots of the different stands, thick solid line for managed laurel forest, slashed lines for laurel forest, dotted grey line for *Pinus radiata* stand and thin grey solid line for *Eucalyptus globulus* stand (Eigenvalues of axes I and II were 0.985 and 0.394 and the cumulative percentage of variance of both axes was 43.0 %). Species names are indicated with the first letter of genus and species: (ab) *Apollonias barbujana,* (ea) *Erica arborea,* (es) *Erica scoporia,* (eg) *Eucalyptus globulus,* (he) *Heberdenia excelsa,* (ic) *Ilex canariensis,* (ip) *Ilex perado,* (pi) *Persea indica,* (pe) *Picconia excelsa,* (pl) *Prunus lusitanica,* (pr) *Pinus radiata,* (la) *Laurus azorica,* (mf) *Myrica faya,* (rg) *Rhamnus glandulosa,* (vm) *Visnea mocanera* and (vt) *Viburnum tinus.*

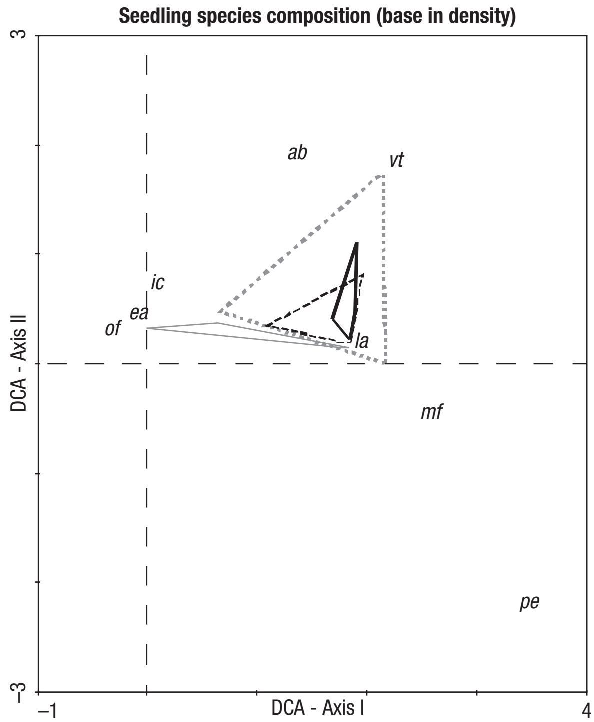

Figure 4. Species scores and plot scores in the ordination space defined by axis I and axis II of the DCA based on density of seedlings. Polygons enclose the plots of the different stands using same lines as in Figure 3 as well as the species names.

individuals that have survived beneath it. Eradication of *P. radiata* by logging can offer good results. However, extra work —and therefore, extra-money— is required for *E. globulus* because the logs must be killed to prevent the abundant and rapid growth of basal sprouts (*i.e.* applying glyphosate on the logs).

We found no evidence to support our preliminary hypothesis that exotic species regenerate under their canopy. Although regeneration differs between exotic and native stands, the results of the study suggest a nurse effect on native species regeneration by these exotic stands and indicate that regeneration is more diverse than canopy composition.

Although the management of those areas is not an urgent issue, we propose some management practices that can be used to convert these plantations to laurel forests and to accelerate the rate of conversion. These measures are related more to restoration than to the invasion control of exotic species. Based on the results of our study, we propose the following management practices:

i) Thinning treatments for the *Pinus radiata* stand. Removing 50% of the trees will favor the heterogeneity of the stand and the establishment of other species, as previously attempted with *P. canariensis* (Arévalo and Fernández-Palacios, 2003). The extremely closed canopy of unmanaged plantations has negative effects on the establishment of shade-tolerant species (Ashton *et al.,* 1998). The ability of natural forest to dominate the stand should be promoted by conducting subsequent stand management activities as thinning of the exotic species (Haggar *et al.,* 1997; Lugo *et al.*, 1993). As long as regeneration of *P. radiata* is very low, the establishment of native vegetation is expected to occur naturally; ii) Total eradication of *Eucalyptus globulus* by logging and post-treatment of the logs, followed by enrichment in the form of plantations of containerized individuals of laurel forest trees, especially the more heliophytic species (*Erica arborea, E. scoparia, Myrica faya* and *Ilex canariensis*). As before, post-treatments should be dependent on the evolution of regeneration.

These relatively expensive practices need particular economic analysis.

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