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Post-fire recovery of the plant community in *Pinus brutia* forests: active *vs.* indirect restoration techniques after salvage logging

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Although reforestation is frequently utilized in many Mediterranean Basin countries to restore burned Mediterranean pine woodlands, post-fire recovery of the plant community is often neglected. To compare the post-fire recovery of the plant community following active and indirect post-fire restoration techniques, we studied three post-fire regeneration treatments in a salvage-logged *Pinus brutia* forest, including two active (plantation and seeding) restoration techniques and one indirect (natural regeneration). An unburned pine stand was also included in the study. We applied the point-intercept method to obtain data on the presence and cover of individual species and functional groups in six replicate one-hectare plots for each treatment. We found no significant differences in plant species richness among post-fire treatments; however, plant community composition and vegetation structure were significantly different between treatments. There was a shift in plant community structure when active restoration techniques were applied, from the woody- and resprouter-dominated plant community of the unburned site to an annual herbaceous- and non-resprouter-dominated one. Our results suggest that active restoration by planting tree saplings in Mediterranean pine forests after a fire may decrease the plant community's resilience and provide empirical evidence that pine plantation treatments change the plant species composition of these forests. These results have important implications for post-fire management of Mediterranean Basin pine forests.

Keywords: Fire, Mediterranean Pine Forest, Plant Cover, Plant Functional Groups, Post-fire Restoration, Resilience, Species Diversity, Turkish Red Pine

Introduction

Pine forests cover large areas throughout the Mediterranean Basin and are of great ecological and economic importance (Scarascia-Mugnozza et al. 2000, Boydak 2004). Turkish red pine (*Pinus brutia* Ten.) and Aleppo pine (*P. halepensis* Mill.) form pine forests with the most prominent crown fire regimes in low altitudes in the Mediterranean Basin (Keeley et al. 2012). In recent decades, especially in the western Mediterranean Basin, socio-economic changes (*e.g.*, the rural-urban migration) and reforestation policies have resulted in homogenous pine stands that are sensitive to drought-driven large wildfires (Pausas et al. 2008, Pausas & Fernández-Muñoz 2012).

Most of the low-altitude Mediterranean pine forests are resilient to fires and successful post-fire regeneration of these forests has been reported by several authors (Thanos & Doussi 2000, Rodrigo et al. 2004, Pausas et al. 2008). However, the failure of post-fire establishment of pine seedlings in these forests has also been recorded, for instance on steep slopes or in poor soils (Thanos & Doussi 2000), and especially if fires are very frequent (Eugenio et al. 2006). Field observations (Baeza et al. 2007, Tessler et al. 2014) and modeling studies (Pausas 1999) suggest that if fires occur more frequently than 20 years, which is the time period needed for pine trees to develop a canopy seed bank (*i.e.*, immaturity risk), then pine forests transform into open shrublands or grasslands.

In low-altitude Mediterranean pine forests, many species of angiosperms have acquired several adaptations to cope with fire (Paula et al. 2009), including resprouting from underground lignotubers and stimulation of germination by heat shock or smoke (Keeley et al. 2011). Owing to these adaptations, post-fire plant diversity increases in the years following a fire until canopy closure is achieved (Keeley et al. 2012) while post-fire successional process also contributes to the gamma diversity of the region as each seral stage has its own specific plant and animal species composition (Tavsanoglu 2008, Kaynas 2017). Many studies have documented a peak in plant diversity during the early stages of post-fire succession in fire-prone Mediterranean Basin habitats (Trabaud & Lepart 1980, Kazanis & Arianoutsou 2004a, Kavgaci et al. 2010).

Post-fire regeneration of Mediterranean pine forests is a major economic concern in many countries within the Mediterranean Basin (Boydak 2004, Pausas et al. 2004a). Therefore, national forest services frequently conduct active restoration by applying different techniques (*e.g.*, pine plantation) for sustainable forestry in these pine forests (Vallejo et al. 2012). Owing to restoration priorities that favor pine plantations, many studies on post-fire regeneration in Mediterranean pine woodlands have also been performed, with a prominent focus on post-fire pine regeneration (Spanos et al. 2000, Tsitsoni et al. 2004). Few studies, however, have addressed the dynamics of plant community structure in burned Mediterranean Basin pine forests (Kazanis & Arianoutsou 2004a, Kavgaci et

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al. 2010). Contrary to the prevailing approach of post-fire management, natural regeneration and seeding methods have been suggested instead of pine plantation to restore burned areas in the Mediterranean Basin successfully and cost effectively (Boydak 2004, Leverkus et al. 2012, Moreira et al. 2012, Vallejo et al. 2012). It has also been suggested that less intensive post-fire management techniques would improve plant diversity in Mediterranean Basin forests (Leverkus et al. 2014), as has been concluded for fire-prone forests in the western United States (Noss et al. 2006). Moreover, owing to global change, post-fire restoration in Mediterranean forests requires a comprehensive approach (Doblas-Miranda et al. 2015). In short, "the restoration of a burned area is not just a matter of how to carry out reforestations" (Moreira et al. 2012).

Turkey is one of the leading Mediterranean Basin countries in reforestation (mostly *Pinus brutia* – Pausas et al. 2008); however, *P. brutia* stands are also utilized intensively for timber production (Ministry of Environment and Forestry of Turkey 2010). Various post-fire management techniques are applied to burned *P. brutia* forests in Turkey (Boydak 2004) without any consideration as to how different restoration techniques affect plant community structure and plant diversity, and with a prominent focus on pine regeneration for future wood production (Ç. Tavsanoglu, personal observation). In Turkey, salvage logging is the first application before any post-fire regeneration treatment in burned *P. brutia* forests due to economic concerns (Akay et al. 2007).

The present study aims to assess the response of the plant community to various post-fire regeneration treatments in salvage-logged *Pinus brutia* forests. We hypothesize that plant diversity differs substantially between active (*i.e.*, planting and seeding) and indirect (*i.e.*, natural regeneration) restoration techniques. We therefore expect more diverse and fire-resilient plant communities in areas subjected to indirect restoration.

Materials and methods

Study area

The study was conducted in Çetibeli district, Marmaris region, south-western Turkey (36.98° N, 28.32° E). The study area is dominated by native Turkish red pine (*Pinus brutia*) forests, with maquis shrubs (such as *Quercus infectoria, Phillyrea latifolia*, and *Myrtus communis*) also found in the understory with high coverage (Tavsanoglu & Gürkan 2014). The Marmaris region is one of the most frequently burned areas in south-western Turkey, with many burned sites co-occurring with unburned forests in the landscape. Consequently, both monopyric (*i.e.*, annuals and post-fire seeders) and polypyric (*i.e.*, post-fire resprouters) species exist in the region as in many typical crown-fire ecosystems (Pausas & Keeley 2014), while the area's vegetation consists of plant communities adapted to crown fire regimes (Tavsanoglu 2008). Stunning examples of monopyric species in southwestern Turkey include *Chaenorhinum rubrifolium*, whose germination is stimulated by multiple fire-related cues (Tavsanoglu et al. 2017), many woody Lamiaceae species that has smoke-stimulated germination (Catav et al. 2014), several *Cistus* species in which germination is stimulated by various heat-shocks, and *Pinus brutia*, a serotinous pine tree (Tavsanoglu & Gürkan 2014). Many large resprouter shrubs, such as *Phillyrea latifolia* and *Arbutus andrachne*, are good examples of the region's polypyric species (Tavsanoglu & Gürkan 2014). The climate is typically Mediterranean, with a prominent summer drought period, and wet winters. Serpentine soils cover most of the study area because of ophiolite rocks formed by underwater volcanic activity in the early Mesozoic. The study area is located between 70 and 400 m above sea level.

Fire and regeneration treatments

In the summer of 2002, a high-intensity, stand-replacing crown fire burned 1775 ha of *Pinus brutia* forest in the study region. After this fire event, fire-killed trees were salvage-logged within the first few months before several regeneration treatments (hereafter "treatment") were applied to the burned area within the first year after the fire by the local forest service (see below). Although the extent of the burned area was sizeable, treatments were applied at a fine scale (in 1-5 ha sites) throughout the burned area.

The treatments applied to the burned area can be classified as active or indirect restoration techniques (Vallejo et al. 2012), which are both frequently applied in the regeneration of salvage-logged *Pinus brutia* forests by forest services (Boydak 2004). The first active restoration technique was the pine plantation (hereafter "plantation") treatment, in which the vegetation remaining after salvage logging was removed by plowing before planting *P. brutia* saplings. The second active restoration technique was the "seeding" treatment, in which, after salvage logging, the cone-bearing branches of the burned trees and the remaining parts of the shrubs were homogeneously placed on the soil surface before additional *Pinus brutia* seeds from seed orchards from a neighboring forest locality were spread by hand over the site. The indirect restoration technique was "natural regeneration", which meant leaving the site as found without applying any treatment after salvage logging.

Study sites and sampling design

We selected the three treatments described above (planting, seeding, and natural regeneration) within the area burned in 2002 and an unburned site next to the burned area. The unburned site had the same characteristics as the burned site in terms of the type of the geological material and soil structure while the pre-fire plant community of the burned site was similar to the unburned site (Tavsanoglu & Gürkan 2010, 2014). Each treatment and unburned site included six 1-ha replicate plots for a total of 24 plots, evenly distributed over three aspects (north, south, and flat; see Tab. S1 and Fig. S1 in Supplementary material). We applied point-intercept methodology to obtain data on the presence and cover of individual species in the selected plots. At a location close to the center of each plot, two line transects, 50 m in length, were established. The distance between two transects in each plot was 20 m. We sampled 100 consecutive points at 50 cm intervals along each transect line, and recorded the plant taxa present at each point to estimate the cover of each species in each transect. This methodology allowed us to estimate mean cover values at the individual taxon level in each plot by averaging the values of the two transects. We also obtained mean cover values for functional groups by summing the cover values of the taxa included in the same functional group. Field sampling was conducted between August 2007 and February 2008. Where possible, plant individuals found at the sampling points were identified to the species level in the field and, if not possible, a specimen was taken for identification at the herbarium. Nomenclature followed Davis (1985), Güner et al. (2012), and Stevens (2001) for the current family names.

Functional groups

We applied regeneration mode and growth form as functional grouping systems to better understand the effects of post-fire treatments on the plant community and vegetation structure at the functional group level. These functional grouping systems are fundamental ones for the fire-prone Mediterranean Basin ecosystems (Tavsanoglu & Gürkan 2014). Regeneration mode grouping was based on the regeneration traits of plants in crown fire ecosystems (Pausas et al. 2004b) and included the following groups: (i) obligate resprouters; (ii) facultative resprouters; (iii) obligate seeders; and (iv) species with no specific post-fire regeneration traits (hereafter "none"). Obligate resprouters regenerate only from under- or above-ground buds by resprouting after fire (R+P- in Pausas et al. 2004b); obligate seeders regenerate only from seeds, have fire-stimulated germination, and lack any resprouting ability (R-P+); and facultative resprouters can regenerate both by resprouting and seeding after fire (R+P+). The none group consisted of non-resprouter species lacking any evidence of post-fire germination stimulation (R-P-). We also considered obligate seeders with a soil seed bank as a separate functional group to distinguish their effects from *Pinus brutia*, which has a canopy seed bank. The regeneration modes of individual taxa were based on Paula et al. (2009), Tavsanoglu & Gürkan (2014), and field observations.

We also applied a binary growth form grouping to distinguish woody and herbaceous species, and these groups were further divided into additional sub-groups (tree, large shrub, shrub, scrub, liana, perennial forb, perennial graminoid, geophyte, annual forb, and annual graminoid) to describe how different growth forms are affected by each treatment. We also analyzed the responses of species belonging to different families to determine if there were any associations between treatments and species taxonomic status.

Data analysis

To obtain the overall cover of the functional group in each plot, we summed the cover of all species in each functional group. Mean cover values of functional groups for each treatment site and the unburned site were obtained by averaging plot values. Therefore, the cover of any functional group could exceed 100%. We also summed the number of species for each functional group. We utilized chi-square analysis to test the statistical significance of the associations between treatments and number of taxa in different functional groups.

Differences in the cover of each taxon and functional group across treatment groups (including the unburned site) were analyzed by one-way analysis of variance (ANOVA). Tukey's HSD *post-hoc* test was applied to analyze the differences between treatments. Two-way ANOVAs were also conducted to test the effects of treatments and aspect on the cover of functional groups. For each functional group, we performed two separate two-way ANOVAs: one included the unburned site and all three post-fire treatments (planting, seeding, and natural regeneration); the other excluded the unburned site. The latter analysis was conducted to reduce the possibility of obtaining overestimated p-values indicating the presence of significant differences among groups, especially considering that the majority of variance in the data came from the unburned site. This enabled us to show more clearly the differences in the cover of functional groups across treatments (without the noise from the unburned site). Cover data were log-transformed before each analysis for a better approximation to the normal distribution.

Permutational multivariate analysis of variance was used to test whether the species composition of the plant community differed across treatments, based on 999 permutations of a Bray-Curtis dissimilarity matrix of the presence/absence of species in study plots. Non-metric multidimensional scaling (NMDS) was also performed to visualize the differences in species composition across treatments. We performed both analyses twice to determine the effects of the unburned site on the overall results: one included the unburned site and all three post-fire treatments (planting, seeding, and natural regeneration); the other excluded the unburned site. For these two analyses, we used "adonis" and "metaMDS" functions in the "vegan" package, respectively (Oksanen et al. 2018).

All analyses were performed using R statistical software (version 3.4.2, R Foundation for Statistical Computing, Vienna, Austria – http://www.R-project.org/).

Tab. 1 - Number of species according to their regeneration mode, growth form, and family in each treatment group. (Natural Reg): natural regeneration.

Group	Subgroup	Unburned	Restoration technique		
			Plantation	Seeding	Natural Reg
All Species	*Total*	*28*	*35*	*40*	*32*
	Obligate resprouters	15	12	11	9
	Facultative resprouters	5	6	7	4
	Obligate seeders	4	8	10	9
	None	4	9	12	10
Resprouters	*Total*	*20*	*18*	*18*	*13*
	Woody	17	11	14	10
	Herbaceous	3	7	4	3
Non-resprouters	*Total*	*8*	*17*	*22*	*19*
	Woody	3	5	5	5
	Herbaceous	5	12	17	14
Woody species	*Total*	*20*	*16*	*19*	*15*
	Trees	2	1	2	1
	Large shrubs	9	3	6	4
	Shrubs	3	5	5	4
	Scrubs	5	6	5	5
	Liana	1	1	1	1
Herbaceous Species	*Total*	*8*	*19*	*14*	*17*
	Perennials	3	8	6	6
	- Perennial forbs	2	2	4	4
	- Perennial graminoids	0	5	2	2
	- Geophytes	1	1	0	0
	Annuals	5	11	15	11
	- Annual forbs	2	6	7	6
	- Annual graminoids	3	5	8	5
Family	Lamiaceae	4	3	3	4
	Fabaceae	1	5	5	3
	Asteraceae	1	3	3	4
	Cistaceae	1	2	2	2
	Poaceae	3	9	10	7

Results

A total of 60 taxa were recorded in the study area (listed in Tab. S2, Supplementary material), belonging to 24 families and 51 genera; however, the study area was dominated by members of the Poaceae, Lamiaceae, and Fabaceae families. Many of the woody taxa were obligate resprouters whereas many of the herbaceous taxa were non-resprouters (Tab. 1). However, there was no relationship between the number of taxa within functional groups (regeneration mode and growth form) and treatments (Tab. 2).

There were significant differences be-

Tab. 2 - Association between treatments and number of taxa in different regeneration modes and different growth forms. The raw data is presented in Tab. 1. (df): degrees of freedom; (RM): regeneration mode; (GF): growth form.

Group	χ^2	df	Prob.
RM (all)	2.0	6	0.916
RM (resprouting/non-resprouting)	6.6	3	0.086
RM × GF (woody/herbaceous)	3.1	6	0.790
GF (all)	17.7	24	0.817
GF (woody/herbaceous)	4.8	3	0.190

Tab. 3 - Results of two-way analysis of variance for the effects of treatment and aspect on the cover of regeneration modes, growth forms, and plant families. Two separate analysis were performed: including and excluding unburned site. (T × A): interaction of treatment and aspect; (ns): non-significant; (*): p < 0.05; (**): p < 0.01; (***): p < 0.001.

Groups	With unburned site			Excluding unburned site		
	Treatment	Aspect	T × A	Treatment	Aspect	T × A
Total	****	*	ns	ns	ns	ns
Obligate resprouters	***	ns	ns	*	ns	ns
Facultative resprouters	ns	ns	ns	*	ns	ns
Obligate seeders	***	ns	ns	ns	ns	ns
Obligate seeders (soil seed bank)	**	ns	ns	ns	ns	ns
None	*	ns	ns	ns	ns	ns
Woody species	****	ns	ns	***	ns	ns
Trees	****	**	**	**	**	*
Large shrubs	****	ns	*	*	ns	ns
Shrubs	***	ns	ns	*	ns	ns
Scrubs	*	*	ns	ns	ns	ns
Liana	*	**	*	ns	**	ns
Herbaceous species	*	ns	ns	*	ns	ns
Perennials	**	ns	ns	ns	ns	ns
- Perennial forbs	ns	ns	ns	ns	ns	ns
- Perennial graminoids	***	ns	ns	*	ns	ns
- Geophytes	ns	ns	ns	ns	ns	ns
Annuals	ns	ns	ns	ns	ns	ns
- Annual forbs	*	ns	ns	*	ns	ns
- Annual graminoids	ns	ns	ns	ns	ns	ns
Lamiaceae	ns	*	ns	ns	ns	ns
Fabaceae	****	ns	ns	ns	ns	ns
Asteraceae	ns	ns	ns	ns	ns	ns
Cistaceae	****	ns	ns	**	ns	ns
Poaceae	**	*	ns	*	ns	ns

Tab. 4 - Mean (± standard error) cover values for each functional group in different treatment sites. The results from the one-way analysis of variance for the effects of treatments on the cover of regeneration modes, growth forms, and plant families are shown. The same letter on the cover values indicates no significant difference (p > 0.05) across treatments. (Natural Reg): natural regeneration.

Groups	Unburned	Restoration technique			F	p
		Plantation	Seeding	Natural Reg		
Total	185.5 ± 14.0 ^{a}	98.7 ± 9.1 ^{b}	100.0 ± 5.8 ^{b}	107.7 ± 6.6 ^{b}	18.0	<0.0001
Obligate resprouters	74.9 ± 16.1 ^{a}	12.8 ± 3.1 ^{b}	18.6 ± 7.5 ^{b}	40.1 ± 6.3 ^{ac}	9.3	0.0005
Facultative resprouters	12.0 ± 7.8	5.4 ± 2.4	13.7 ± 1.2	9.6 ± 2.0	2.0	0.141
Obligate seeders	93.7 ± 2.4 ^{a}	39.8 ± 3.9 ^{b}	59.8 ± 7.5 ^{c}	52.9 ± 5.6 ^{bc}	14.3	<0.0001
Obligate seeders (soil seed bank)	5.5 ± 2.9 ^{a}	31.8 ± 4.5 ^{b}	57.2 ± 7.0 ^{c}	51.5 ± 5.5 ^{bc}	20.2	<0.0001
None	4.9 ± 2.4 ^{a}	40.7 ± 11.2 ^{b}	7.9 ± 2.6 ^{ab}	5.1 ± 2.4 ^{ab}	3.7	0.029
Woody species	114.7 ± 6.1 ^{a}	37.2 ± 6.8 ^{b}	75.2 ± 7.2 ^{c}	64.5 ± 4.8 ^{c}	22.4	<0.0001
Trees	88.5 ± 2.3 ^{a}	8.0 ± 2.6 ^{b}	2.7 ± 0.9 ^{b}	1.4 ± 0.4 ^{b}	20.7	<0.0001
Large shrubs	62.2 ± 10.3 ^{a}	5.7 ± 2.0 ^{b}	11.2 ± 5.5 ^{b}	16.5 ± 4.0 ^{b}	12.8	<0.0001
Shrubs	12.8 ± 7.7 ^{a}	23.0 ± 3.2 ^{b}	50.4 ± 3.4 ^{b}	44.8 ± 8.0 ^{b}	11.6	0.0001
Scrubs	2.3 ± 1.0	5.7 ± 3.4	18.7 ± 7.7	15.6 ± 8.1	2.3	0.109
Liana	11.1 ± 5.0	0.5 ± 0.3	3.5 ± 2.8	2.7 ± 2.2	2.0	0.140
Herbaceous species	8.7 ± 3.3 ^{a}	55.8 ± 12.4 ^{b}	13.7 ± 6.4 ^{a}	26.7 ± 3.8 ^{ab}	8.1	0.001
Perennials	0.7 ± 0.5 ^{a}	6.5 ± 2.6 ^{bc}	4.8 ± 3.0 ^{ac}	18.8 ± 2.7 ^{b}	10.1	0.0003
- Perennial forbs	0.6 ± 0.5	0.2 ± 0.1	1.2 ± 0.4	1.2 ± 0.7	1.4	0.279
- Perennial graminoids	0 ^{a}	5.9 ± 2.8 ^{bc}	3.6 ± 2.8 ^{ab}	17.7 ± 3.0 ^{c}	14.0	<0.0001
- Geophytes	0.1 ± 0.1	0.1 ± 0.1	0	0	0.7	0.582
Annuals	8.0 ± 3.5	49.3 ± 12.4	8.9 ± 3.7	7.8 ± 3.8	2.4	0.093
- Annual forbs	3.5 ± 2.3 ^{ab}	12.8 ± 3.5 ^{b}	1.8 ± 1.4 ^{a}	2.4 ± 1.6 ^{ab}	3.4	0.038
- Annual graminoids	4.5 ± 2.3	36.5 ± 10.5	7.0 ± 2.4	5.4 ± 2.5	3.0	0.057
Lamiaceae	4.5 ± 2.8	1.7 ± 0.8	1.1 ± 0.5	0.6 ± 0.3	0.7	0.584
Fabaceae	0.2 ± 0.2 ^{a}	13.8 ± 4.6 ^{b}	23.9 ± 5.5 ^{b}	21.8 ± 6.2 ^{b}	57.4	<0.0001
Asteraceae	0.1 ± 0.1	1.5 ± 1.1	0.8 ± 0.4	1.0 ± 0.5	1.6	0.219
Cistaceae	2.3 ± 0.6 ^{a}	18.0 ± 1.0 ^{b}	39.1 ± 3.9 ^{c}	35.9 ± 6.3 ^{bc}	51.4	<0.0001
Poaceae	4.5 ± 2.3 ^{a}	41.3 ± 10.2 ^{b}	10.6 ± 4.9 ^{ab}	23.1 ± 2.6 ^{b}	7.8	0.001

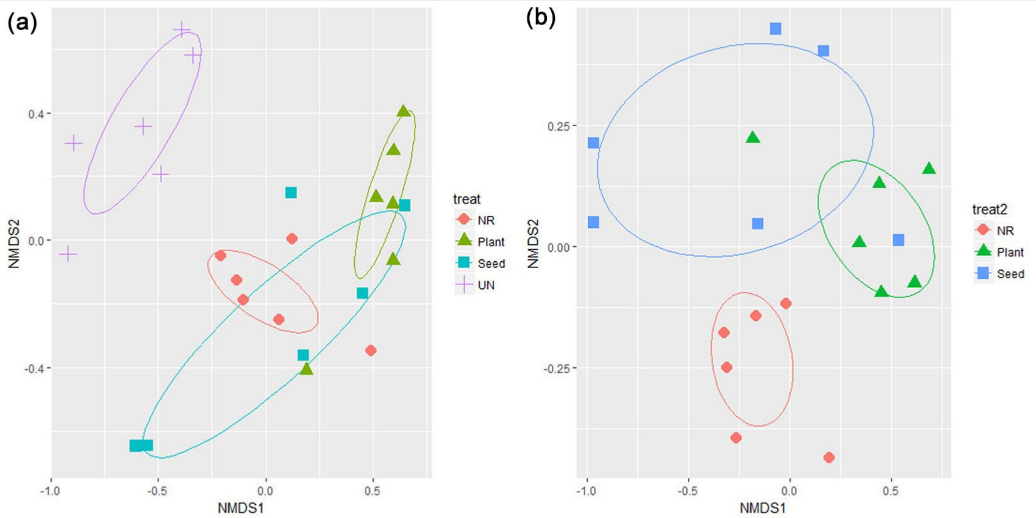

Fig. 1 - NMDS ordination graph of species presence across treatments. Two graphs are given: including (a) and excluding (b) the unburned site (UN) from the analysis (see Materials and methods section for details). Each point represents the species composition of the plant community in a single 1-ha plot. Confidence ellipses are also shown for each treatment. (NR): natural regeneration; (Seed): seeding; (Plant): plantation.

tween the unburned site and treatment sites in the cover of one-third of the studied taxa (Tab. S3 in Supplementary material). Most differences were related to the regeneration mode of the taxa. That is, cover of seeder taxa, such as *Cistus* spp. and *Cytisopsis pseudocytisus* ssp. *reeseana*, were significantly higher in the seeding and natural regeneration treatments than in the unburned site. Cover of dominant resprouters either decreased (*Quercus infectoria, Myrtus communis*, and *Styrax officinalis*) or remained unchanged (*Phillyrea latifolia*) in burned sites compared to the unburned site. Significant increases in the cover of taxa in the none group were observed in the plantation treatment (*e.g., Bupleurum orientale, Avena fatua*, and *Bromus* spp. – see Tab. S3 in Supplementary material).

The results of the two-way ANOVAs showed that the unburned site was clearly different from the burned sites regarding the cover of functional groups (Tab. 3). By excluding the unburned site, less significant results were obtained in almost all functional groups, although the effects of the treatments were still significant for some functional groups. This analysis also showed that different post-fire treatments resulted in different cover values only for resprouters, woody species, and trees (Tab. 3). Aspect made a relatively limited contribution to the total variance and had a significant effect only on the cover of trees and liana (Tab. 3).

More detailed analysis of the cover of functional groups in different treatments indicated that there were significant increases or decreases in different treatment groups compared to the unburned site (Tab. 4). Considering the woody species, the total cover of trees (mainly *Pinus brutia*) was dramatically reduced in the burned sites compared to the unburned site (from 88% down to 1%) while that of shrubs and scrubs increased (from 13% up to 50% and from 2% up to 19%, respectively). Total cover of herbaceous species significantly increased only in the plantation treatment compared to the unburnt site (from 9% to 56%) whereas no significant changes were found in either the seeding or natural regeneration treatments (Tab. 4). Increase in the cover of annuals was responsible for the significant change in total herbaceous cover in the plantation treatment; however, the lack of any increase in the cover of annuals resulted in an insignificant increase in the seeding and natural regeneration treatments. Specifically, the cover of perennial graminoids was significantly higher in the natural regeneration treatment than in the unburnt site (18% and 0%, respectively). The cover of Cistaceae, Poaceae, and Fabaceae families was significantly higher in the burned sites than in the unburned site whereas no significant change was observed in the cover of Lamiaceae and Asteraceae families.

NMDS of species presence data differentiated the unburned site from burned sites (*i.e.*, treatment groups – Fig. 1a), as was confirmed by the permutational multivariate analysis of variance (F = 5.6, R^2 = 0.46, p = 0.001). Moreover, after excluding the unburned site, the cross-treatment differences were still significant (F = 3.4, R^2 = 0.31, p = 0.003) and NMDS analysis revealed three separate groups (Fig. 1b). However, a clear difference in the two-dimensional space of the NMDS graph appeared between indirect (*i.e.*, natural regeneration) and active (*i.e.*, seeding and planting) restoration treatments after excluding the unburned site from the analysis (Fig. 1b). These results indicate a change in community composition in the burned sites compared to the unburned site. Moreover, they show that the natural regeneration treatment sites had different plant species compositions to the plantation and seeding treatment sites.

Discussion

Our study showed that plant community composition and vegetation structure significantly changed in burned sites compared to the unburned site in *Pinus brutia* forests. Moreover, this change became more drastic in areas where the plantation treatment was applied. Considering the coverage of plant functional groups, seeding and natural regeneration treatments had less impact on the vegetation structure than the plantation treatment did in *P. brutia* forests after fire.

Short-term increases in the number of plant species in burned pine forests of the Mediterranean Basin is very common (Kazanis & Arianoutsou 2004a, Kavgaci et al. 2010), which results from fire adaptations observed in many plants (Paula et al. 2009). Management decisions in favor of recruiting pine plantations, however, may lead to the disruption in the natural recovery process in such forests (Maestre & Cortina 2004, Gómez-Aparicio et al. 2009). On the contrary, it is reported that plowing has minor and temporary effects on plant species richness and vegetation cover; however, it negatively affects pine regeneration in Mediterranean pine woodlands (Hibsher et al. 2013). As the benefits of planting seedlings offset by its negative effect on the small mammal community in a pine woodland dominated by the non-serotinous species *Pinus nigra*, it has been suggested that a multi-criteria approach should be implemented before selecting post-fire management practices in such forests (Espelta et al. 2003). In our study, although there were no significant differences in plant species richness across the post-fire treatments, plant community composition and vegetation structure for natural regeneration and the other two (seeding and plantation) restoration techniques differed significantly. Consequently, many of the plants found in the seeding and plantation areas were not found in the natural regeneration areas and *vice versa*. This result indicates that there was a shift in plant community structure when active restoration techniques were applied, specifically from the woody- and resprouter-dominated plant community of the unburned site to an annual herbaceous- and non-resprouter-dominated one. Consequently, we rejected our first hypothesis that plant diversity differs across treatments. However, our results supported the second hypothesis – that indirect restoration sites have a more fire-resilient community than active ones. This result also shows that such a shift in the plant community composition can be masked if only species diversity measures are considered (Moya et al. 2009, Alfaro-Sánchez et al. 2015). The case mentioned above was more dramatic in the plantation treatment in which annual species had approximately 40% more cover than others. The active restoration treatments we included in our study also differed from each other in plant species composition and the cover of the seeder functional group in that both were lower in the plantation than the seeding treatment. This difference suggests that fire-adapted seeder species were eliminated in the plantation treatment and replaced by non-resprouter species without any adaptation to fire. On the other hand, tree (*i.e.*, pine) cover in the plantation treatment was six- and three-times higher than in the natural regeneration and seeding treatments, respectively. This result indicates that plantation is a successful treatment to regenerate the target species *Pinus brutia* in burned pine woodlands in areas where wood production is the main ecosystem service (Ozkan & Ozdemir 2016). However, the negative effects of pine plantations on several ecosystem services, such as providing suitable animal habitat, preventing soil erosion, and maintaining biodiversity, have also been acknowledged (Maestre & Cortina 2004, Ozkan & Ozdemir 2016). Even if the only management goal is regenerating the dominant pine species of the plant community in some areas, biodiversity is still an important part of these ecosystems, which provides services like promoting forest resilience (Thompson et al. 2009). In some cases, moreover, even salvage logging is not the best option for post-fire management and it could have long-term negative impacts on ecological and socio-economical services (Castro et al. 2011). Unfortunately, because our sampling design did not include plots of natural regeneration without salvage logging, we cannot evaluate the possible effects of salvage logging on the post-fire recovery of plant community. Therefore, future studies about post-fire management in *P. brutia* forests should include burned plots in which salvage logging is not applied for a better understanding of the effects of salvage logging on the plant community in such forests.

Resilience is high in resprouter-dominated vegetation in fire-prone Mediterranean Basin areas (Keeley 1986, Díaz-Delgado et al. 2002, Baeza et al. 2007, Pausas et al. 2008, Granados et al. 2016) since resprouters can recover quickly after fire. Post-fire resilience in pine-dominated plant communities is also dependent on fire-persistent understory vegetation, including both resprouters (Díaz-Delgado et al. 2002) and seeders (Arnan et al. 2007). Consequently, resilience to disturbance is a key functional element of Mediterranean plant communities (Keeley 1986, Lavorel 1999), and a deviation from the plant community composition that can be expected from an autosuccessional model (*i.e.*, direct regeneration) is a potential indicator of low resilience in Mediterranean pine forests (Kazanis & Arianoutsou 2004b). The apparent change in plant community structure in our study, as indicated by significantly lower cover values of resprouters and seeders in plantation sites, therefore suggests that active restoration by planting Mediterranean pine forests after fire (i.e. ploughing and then replanting pine saplings) may decrease the resilience of the plant community compared to less artificial restoration tools (*i.e.*, seeding and natural regeneration in our case). Furthermore, large shrubs had a significant place in the plant community in both the unburned and naturally-regenerated sites in our study, which suggests that these species are sensitive to active restoration techniques. As one of the providers of high forest resilience, this functional group could be used as a substitute for the tree component of the community for ecosystem services such as wood production (*i.e.*, coppicing) and soil protection.

Successful pine regeneration can be achieved through indirect restoration techniques in burned Mediterranean pine forests (Spanos et al. 2000, Pausas et al. 2004a, Tsitsoni et al. 2004). No effect of using log and branch barriers has been demonstrated in post-fire regeneration in Mediterranean pine forests (Raftoyannis & Spanos 2005) whereas the existence of branches on the forest floor is reported to have positive effects (Eron & Sarigül 1992, Pausas et al. 2004c, Castro et al. 2011). In this respect, our results support the suggestion that indirect (*i.e.*, natural regeneration) and relatively less artificial (*i.e.*, seeding) restoration techniques should be applied instead of plantation in many cases while restoring burned fire-prone Mediterranean pine forests (Neeman 1997, Pérez & Moreno 1998, Boydak 2004, Maestre & Cortina 2004, De las Heras et al. 2012, Vallejo et al. 2012, Marañón-Jiménez et al. 2013, Leverkus et al. 2014). The functional group approach we used in our study provided practical knowledge to improve resilience in Mediterranean pine forest ecosystems and may help to predict the outcomes of future changes in fire regimes to which the plant community is adapted.

Conclusions

Our results provide empirical evidence that plantations are the least beneficial treatment for resilience in Mediterranean pine forests (Maestre & Cortina 2004, Jucker Riva et al. 2016). We therefore recommend that seeding and natural regeneration techniques should be preferred in salvage-logged *Pinus brutia* forests after fire. On the other hand, other restoration options that we did not test in our study may also help to improve the diversity and resilience of post-fire plant community, such as the plantation of resprouter species (Granados et al. 2016) and pine planting without removing regenerating vegetation. Our findings on the plant community shift also indicate that plant functional groups provide an effective tool for evaluating the effects of post-fire management approaches in fire-prone ecosystems (Bradstock & Kenny 2003). The results of our study have important implications for post-fire management of Mediterranean Basin pine forests.

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Supplementary Material

Fig. S1 - Location of plots in the study area and location of study area in south-western Turkey.

Tab. S1 - Geographic locations, altitudes, and aspects of study plots.

Tab. S2 - Families, growth forms, and regeneration modes of the taxa considered in the study.

Tab. S3 - Mean (± SE) cover values of each taxon considered in the present study in different treatment sites.

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