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Reforestation of degraded Kermes oak shrublands with planted pines: effects on vegetation cover, species diversity and community structure

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Abstract This paper examines the results of plantings of the Mediterranean pine species, *Pinus halepensis* and *Pinus pinea*, in a degraded Mediterranean kermes oak (*Quercus coccifera)* shrubland in Northern Greece, which were accomplished in order to mitigate ecosystem degradation. Plant establishment and the vegetation differences between the degraded ecosystem's previous state and the new state following reforestation were measured in order to evaluate the effect of reforestation. Monitoring of the seedling survival and growth of the planted species was carried out during the next five years. In the fifth year we conducted botanical inventories in 18 and 15 plots (50 m^2 in size) from the reforested and control area, respectively. Plant community parameters estimated were: vegetation composition, total plant cover, planted species cover, native woody, herb and grass species cover, plant species richness, Shannon-Weiner index, community structure and dominant plant height. *P. halepensis* exhibited higher survival and growth than *P. pinea*. The reforested area exhibited higher plant diversity, higher vegetation cover, taller plants and more complex community structures than the control area, which concludes that plantings of pines can be successfully used in degraded ecosystem reforestation projects, in areas with similar site conditions.

Keywords Diversity · *Pinus halepensis* · *Pinus pinea* · Plantings · Reforestation · Species richness

Introduction

Ecological restoration can be defined as ''the process of assisting the recovery of an ecosystem that has been degraded, damaged or destroyed'' (SER 2004). Assisting ecosystem recovery augments biodiversity and ecosystem services, at the landscape scale (Aronson et al. 2006). Restoration of degraded ecosystems is of high importance in

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Mediterranean countries where ecosystem degradation has a long history (Perez-Trejo 1994). Degradation usually appears in areas of low elevation which humans have colonized for centuries (Thirgood 1981; Maestre and Cortina 2004). In Mediterranean lands, where people so firmly inhabit the landscape and manipulate so intensively the flora and fauna of land, air and water (Naveh 1990), a complete restoration of natural ecosystems seems particularly unobtainable (Aronson et al. 1993).

Recently, the reforestation contribution to accelerate natural succession and recover late-successional vegetation has been recognized (Honnay et al. 2002), and specifically the restoration of degraded lands by the reintroduction of woody species, has become increasingly important worldwide (Keenan et al. 1997; Lugo 1997; Carnevale and Montagnini 2002; Maestre and Cortina 2004). Reforestation benefits may include the enhancement of biodiversity, ecosystem stability, protection from soil erosion, and increased carbon sequestration (Maestre and Cortina 2004). This consideration is more obvious in the Mediterranean basin where many reforestation efforts do not include the potential economic value of wood products, but they usually aim at the above-mentioned values (Hatzistathis and Dafis 1989).

Degraded Mediterranean kermes oak shrublands occupy extensive areas in Greece and they so far are characterized by heavy degradation due to the above-mentioned long-time human activities, mainly repeated fires and overgrazing (Baeza et al. 2007). These shrublands are characterized by low vegetation cover and low plant height (commonly less than 2 m), and they are dominated by the evergreen oak species *Quercus coccifera,* accompanied with *Phillyrea latifolia* and *Quercus pubescens* (Dafis 1973). The recovery of these ecosystems is of high importance and can be carried out either naturally (as a result of secondary succession) or by artificial interventions (Tsitsoni et al. 1999). Artificial intervention, by planting pines is a common rehabilitation tool in these ecosystems. Unfortunately, data concerning the results of the above-mentioned recovery methods in these ecosystems are still missing, especially in Greece (Zagas et al. 1998).

Thus, the aim of this paper was to evaluate the results of reforestation in a degraded Mediterranean kermes oak shrubland, with plantings of Mediterranean pine species *Pinus halepensis* Miller (Aleppo pine) and *Pinus pinea* L. (Stone pine) by the Forest Service. In order to evaluate the reforestation effect on these ecosystems, plant establishment and vegetation differences between the degraded area subjected to reforestation and of the same degraded area with no human intervention were estimated.

Materials and methods

Study area

The experimental area is located at the Livadi-Vassilika, North Greece, 28 km far from the city of Thessaloniki. This area is degraded public land used for grazing and covered by low shrubs subjected to overgrazing. Forests are absent and the natural vegetation of the general area is dominated by the evergreen oak species *Quercus coccifera,* accompanied with *Phillyrea latifolia*, *Quercus pubescens* and other woody species (Dafis 1973). The altitude ranges from 200 to 300 m asl, and the slope (generally south faced) varies between 15 and 30%. The geological bedrock belongs to magmatic series of Chortiatis and consists of green schists and epigneisses. The soil is shallow with many erosion problems. The climate is Mediterranean with a 5-month dry period (where the temperature is twice the precipitation) and mean annual temperature 15.6°C and mean annual rainfall 416 mm, according to the nearest meteorological station of Loutra Thermi, 8 km far. However, the maximum air temperatures during the studied period reached 42.8°C in the summer of 2000, while the minimum air temperatures were −11.1°C (in the winter of 2001). Annual rainfall exhibited great fluctuation during the studied period with a minimum value 257 mm (in 2000) and a maximum value 730 mm (in 2002).

In the study area, the Forest Service carried out plantings during the winter in 1998 following mechanical site preparation (sub-soiling, approximately 30 cm in depth, was applied before planting by a Caterpillar D8). Two native pine species *Pinus halepensis* and *Pinus pinea* that are characterized as early successional species, were planted as 2-year old paper pot seedlings on 2 × 3 m spacing in three different locations. Planting holes were 30 cm in diameter and depth.

Plant species sampling and measurements

In this study, three blocks sized 0.12 ha each were established, each of them in one of the above three locations, spaced approximately 1 km. Within each block, 6 permanent plots of 200 m^2 were established (3 plots per species, randomly distributed). Each plot included 30 planted pine seedlings that were selected for field data sampling. All seedlings were numbered and their shoot height and root collar diameter were measured prior to planting. Monitoring of seedling survival and growth of *P. halepensis* and *P. pinea* was carried out once a year (in autumn) for five successive years. Furthermore, in the 5th year we measured the crown dimensions of the seedlings.

An overall evaluation of the reforested area was carried out in the 5th year when, except for the survival and growth of planted pine seedlings, we measured the following plant community parameters: species composition, total plant cover, planted species cover, native woody, herb and grass species cover (according to Domin scale of cover/abundance), plant species richness, Shannon-Weiner index, community structure and dominant plant height. Species richness is simply a count of the number of plant species present in a sample plot (Magurran 1988; Onaindia and Mitxelena 2009) and, when used in combination with diversity indices, can reveal other trends in plant diversity that the more complex indices may not reveal. The Shannon-Wiener index of diversity was calculated by the following equation (Magurran 1988):

$$H = -\sum_{i=1}^{s} (\mathrm{p}i)(\mathrm{Ln}\,\mathrm{p}i)$$

where H = index of species diversity, s = Number of species, pi = proportion of total sample belonging to the ith species.

All the above parameters were estimated both in the reforested area (within the three blocks) as well as in the adjacent degraded area that was used as control, at the same time (5 years after the plantings), in order to evaluate the effect of reforestation (Hobbs and Norton 1996; Yirdaw 2001; Atsamo 2002). Measurements were carried out in 50 m^2 subplots, which were randomly established within each planted plot (a total of 18 subplots in the reforested area) and in another 5 subplots of 50 m^2 that were established in the adjacent control area of each block at a distance of 100 m (a total of 15 subplots in the control area). The size of the subplots was based on species-area curves taken from the literature for Mediterranean type ecosystems (Lunt 1990). For all areas, 33 subplots of 50 m^2 were examined, and at each of these, all the plant species were recorded. Species were classified as native and exotic. Nomenclature follows to Flora Europaea

(Tutin et al. 1993). However, only for the estimation of the herb and grass species cover, five subplots (2 × 2 m) were randomly selected within each subplot of 50 m^2 (Rebele and Lehmann 2002). Also, mean dominant height was estimated by measuring the five tallest individuals of each plant species per subplot (Buford and Burkhart 1987).

Data analysis

Statistical analysis was performed using the SPSS package, version 11.5 for windows. The percentages were transformed to arsine square root values before analysis to normalize the distribution. Paired *t* tests were used for the comparison between the two planted species survival and morphology during the 5 successive years. Student's *t* test procedure was used to detect significant differences in vegetation features between the reforested and control area (Norusis 1994; Fahnestock and Detling 1999; Rebele and Lehmann 2002).

Results

Planted species performance

Analyzing the survival rate of the planted species 5 years after plantation establishment (Fig. 1), we found that *P. halepensis* was a more successful than *P. pinea*, as the former exhibited a significantly higher survival rate (81.8%) than the latter (70.3%). Seedling mortality was observed mainly during the first 2 years after planting and especially during the first summer due to severe drought. After the 2nd year, very low mortality was observed for both species despite the difficult weather conditions occurring during that period; low annual precipitation (257 mm in 2000) and extreme temperatures (42.8°C absolute maximum and −11.1°C absolute minimum).

Average annual height and diameter growth was greater for *P. halepensis* seedlings (31.1 cm and 8.8 mm respectively) than *P. pinea* (12.7 and 7.0 mm, respectively) (Table 1). Five years after planting *P. halepensis* trees were significantly taller (194 cm) than *P. pinea* (99 cm). A great difference between the species was in their ground cover; *P. halepensis* contributed to an average of 26.6% ground cover, while *P. pinea* has significantly lower contribution (10.3%).

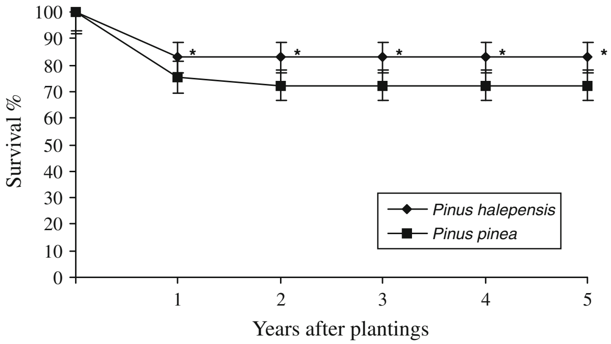

Fig. 1 Survival of planted pines during the 5 years after the plantings. Significant differences ($P < 0.05$, Paired *t* test) in each date is denoted by an *asterisk*. *Vertical bars* represent one standard error of the mean

Table 1 Morphological characteristics of seedlings at planting and 5 years later in the restored area

Species	Initial height (cm)	Initial diameter (mm)	5-year shoot height (cm)	5-year diameter (mm)	Mean annual height growth (cm)	Mean annual diameter growth (mm)	5-year Dimensions of crown (cm)		Canopy cover (%)
							Length (cm)	Width (cm)	
P. halepensis	38.3* (1.6)	5.1 (0.1)	194.1* (6.0)	49.3* (1.2)	31.1* (1.3)	8.8* (0.2)	135.5* (3.6)	144.9* (13.2)	26.6* (5.6)
P. pinea	35.4 (1.4)	5.0 (0.1)	99.2 (2.0)	40.0 (1.1)	12.7 (0.9)	7.0 (0.2)	82.8 (2.2)	81.3 (2.4)	10.3 (3.2)

Values in brackets represent one standard error of the mean

Values in the same column followed by an asterisk are significantly different ($P < 0.05$, Paired *t* test)

Changes in vegetation composition and community structure

The floristic composition in both communities (reforested area and control) was quite similar. The total number of plant species recorded in all plots in both communities was 96 (Table 2). However, the average number of species recorded in each plot was significantly higher in the reforested area compared to control area (47.4 and 41.8, respectively—Table 3), due to the two planted species, and a few species such as *Globularia alypum L.* that were recorded only in the reforested area. The number of woody species was also higher in the reforested area compared to the control (9 and 7 species per plot, respectively). The Shannon-Wiener index was also significantly greater in the reforested area (2.98) than in the control area (2.60). Also, some differences in species abundance were observed between the reforested and control area. For example, *Chrysopogon gryllus (L.)* Trin. was the dominant grass species in the reforested area where it comprised 18.1% of the total plant cover, but it was less abundant in the control area (6.0%) (Fig. 2). In contrast, *Cynodon dactylon (L.)* Pers. was a common dominant species in the control area (20.3%) while it comprised a lower percentage in the reforested area (5.1%). Only one species, *Solanum elaeagnifolium* Cav. was exotic (alien species), while the rest belong to the native flora. Vegetation height was greater in the reforested area not only due to the presence of the planted pine species, but also to the greater height of the native species (Table 3). The mean dominant height of *Quercus coccifera* was 126 and 112 cm in the reforested and control area, respectively. Similar results were observed for the main grass species *Chrysopogon gryllus* and *Cynodon dactylon*; their mean dominant height reached 115 cm in the reforested area and 84 cm in the control.

The reforested area exhibited higher vegetation cover than the control area (Fig. 2); the average cover across all the reforested area was 81.5%, while in the control area it was 76.3%. Much of this increase can be attributed to the cover of the planted species. By far, the most common and dominant species found in all cases was *Quercus coccifera* L. which covered 17.0%, on average, in the reforested area and 24.6% in control, followed by *Phillyrea latifolia* L. and *Anthyllis hermanniae* L.*,* while the rest common woody species *Erica manipuliflora* Salisb.*, Juniperus oxycedrus* L.*, Cistus incanus* L. had a low percentage in all cases (below of 5.0% of the total cover). Cover of herbs and grasses was consistently the same in the reforested and control areas, ranging from 40.3 to 37.3%, respectively.

Discussion

Although, the flora was quite similar in both communities, species diversity and cover were significantly higher in the reforested area than in the control. The vegetation cover in the first case ranged from 75 to 90%, while in the last from 60 to 80%. Also, the vegetation height was significantly taller in the reforested than in the control area. Vertical structure was also improved in the reforested area, since the planted trees in combination with the tallest native shrubs (*Q. coccifera* and, in some cases, *P. latifolia*) contributed to the formation of a two-storey community with an overstorey of the above-mentioned species and an understorey of the rest species. This more complex vertical structure usually favours biodiversity (Pitkänen 1997; Kerr 1999; Ferris et al. 2000). Another important difference was the seed production observed in many species in the reforested area (data not shown); it was observed that many species both natural and planted (especially *Q. coccifera* and *P. halepensis*) produced many seeds, which could explain the higher diversity of the

Table 2 List of plant species found in the restored and control areas

	Plant species cover Domin scale*			Plant species cover Domin scale	
	Restored area	Control area		Restored area	Control area
Woody species			Herb and grass species		
Pinus halepensis	6	–	*Bromus japonicus*	1	1
Pinus pinea	4	–	*Silene italica*	1	1
Quercus coccifera	5	5	*Thymus vulgaris*	1	1
Phillyrea latifolia	3	3	*Lamium amplexicaule*	1	1
Anthyllis hermaniae	3	3	*Phleum montanum*	1	1
Cistus incanus	3	3	*Tragopogon pratensis*	1	1
Juniperus oxycedrus	3	3	*Muscari neglectum*	1	1
Erica manipuliflora	2	2	*Arabis sagittata*	1	1
Solanum eleagnifolium	2	1	*Euphorbia apios*	1	1
Globullaria alypum	2	–	*Genista carinalis*	1	1
Asparagus acutifolius	2	2	*Dactylis glomerata*	1	1
Pyrus amygdaliformis	1	1	*Allium neapolitanum*	1	1
Calicotome villosa	1	1	*Alllium sphaerocephalum*	1	1
Dorycnum hirsutum	1	1	*Carlina vulgaris*	1	1
Erica arborea	1	1	*Scabiosa tenuifolia*	1	–
			Aliaria petiolata	1	1
Herb and grass species			*Echium plantagineum*	1	1
Chrysopogon gryllus	5	4	*Catananche caerulea*	1	1
Cynodon dactylon	4	5	*Tanacetum corymbosum*	1	1
Thymus sibthorpii	3	3	*Centaurea cyanus*	1	1
Melica ciliata	3	3	*Borago officinalis*	–	1
Aegilops neglecta	2	2	*Catapodium rigidum*	1	1
Cynosurus echinatus	1	1	*Picris hieracioides*	1	1
Avena sterilis	1	1	*Lactuca virosa*	–	1
Chenopodium album	1	2	*Agropyron repens*	1	–
Poa nemoralis	1	1	*Micromeria juliana*	1	1
Stipa bromoides	1	1	*Satureja vulgaris*	1	1
Lagurus ovatus	1	1	*Lactuca viminea*	1	1
Teucrium chamaedrys	1	1	*Hordeum bulbosum*	1	1
Bromus squarrosus	1	1	*Silene italica*	1	1
Poa bulbosa	1	1	*Euphorbia helioscopia*	1	1
Centaurea affinis	1	1	*Dianthus cruentus*	1	1
Taraxacum officinale	1	1	*Veronica chamaedrys*	1	1
Briza media	1	1	*Armeria canescens*	1	1
Marubium peregrinum	1	2	*Polygonum arenarium*	1	1
Portulaca oleracea	1	1	*Alyssum chalcidicum*	1	1
Polygala vulgaris	1	1	*Nigella damascea*	1	1
Erodium cicotarium	1	1	*Senecio vulgaris*	–	1
Cerastium sp.	1	1	*Aira elegantissima*	1	1

Table 2 continued

	Plant species cover Domin scale*			Plant species cover Domin scale	
	Restored area	Control area		Restored area	Control area
Crupina vulgaris	1	1	*Cyclamen hederifolium*	1	–
Crupina crupinastrum	1	1	*Teucrium capitatum*	1	1
Cichorium indybus	1	1	*Stellaria media*	1	1
Gnaphalium sylvaticum	1	1	*Galium verum*	1	1
Artemisia vulgaris	1	1	*Festuca valesiana*	1	1
Tordylium officinale	1	1	*Carex flacca*	1	1
Lunaria annua	1	1	*Vicia hirsuta*	1	1
Silene vulgaris	1	1	*Hippocrepis emerus*	1	1
Leondoton hispidus	1	1			
Plantago lanceolata	1	1			
Myosotis arvensis	1	1			
Centaurea solstitialis	1	1			

* Domin scale was recorded as follows: Cover 91–100% as Domin 10, Cover 76–90% as Domin 9, Cover 51–75% as Domin 8, Cover 34–50% as Domin 7, Cover 26–33% as Domin 6, Cover 11–25% as Domin 5, Cover 4–10% as Domin 4, Cover <4% with many individuals as Domin 3, Cover <4% with several individuals as Domin 2, and Cover <4% with few individuals as Domin 1

Table 3 Plant species diversity and dominant height in the restored (5 years after planting) and control areas; n = 18 and 15, respectively

Vegetation characteristics	Restored area	Control
Species richness	47.4 (0.8)*	41.8 (1.2)
Number of woody species	9.0 (0.6)*	7.0 (0.4)
Shannon–Wiener index	2.98 (0.04)*	2.60 (0.09)
Mean dominant height of planted species (cm)	169.0 (9.0)	–
Mean height of the dominant native woody species *Quercus coccifera* (cm)	126.0 (2.0)*	112.0 (2.0)
Mean height of the dominant grass *Chrysopogon gryllus* and *Cynodon dactylon* (cm)	115.0 (2.0)*	84.0 (2.0)

Values in brackets represent one standard error of the mean. Within a row, significant differences ($P < 0.05$, *t* test) are denoted by an asterisk

ecosystems as they comprise a primary food source for many fauna species (Kerr 1999). The self-renewing of restored communities is an important goal for many restoration projects (Huston 1998). Furthermore, acorn production by *Q. coccifera* comprises the basis of many food webs. An early cone production of planted *P. halepensis* saplings was also recorded by Zagas et al. (2004).

Both planted species had high survival and growth rates, particularly *P. halepensis*. The mechanical site preparation may favour seedlings survival of both species, mainly by loosening the soil and thus, allowing an easier expansion of the seedlings root system to soil (Hatzistathis et al. 2000). The first 2 years are critical for pine survival when high

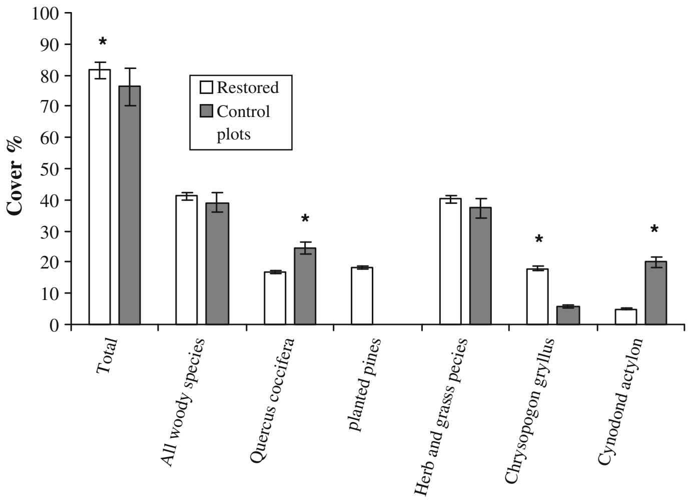

Fig. 2 Vegetation cover (%) in the restored (5 years after the planting) and control plots. Significant diferrences ($P < 0.05$, t test) between the studied plots, are denoted by an asterisk. Vertical bars represent one standard error of the mean

mortality is observed. The higher survival percentage of *P. halepensis* shows that this species has higher plasticity than *P. pinea* and it can also successfully cope with low temperatures such as those that prevailed during the study. Similar results were reported by Zagas and Ganatsas (2001). Both results show that *P. halepensis* presents the best all-round performance under the conditions of degraded Mediterranean kermes oak shrubland. Vidacovic et al. (1990) also reported the same results in Croatia (South Europe) at similar climatic conditions where *P. halepensis* and *P. brutia* were the most successful planted species.

Changes in vegetation characteristics showed that the reforested area exhibited higher plant diversity, higher vegetation cover, taller plants and more complex community structures than the control area. Vegetation changes after plantations with woody species were also recorded by others (Yirdaw 2001; Atsamo 2002); however, the changes depend upon the native vegetation and the planted species as well (Augusto et al. 2001).

Habitat invasion by exotics is another point for evaluating the effect of reforestation. *Solanum elaeagnifolium*, the only exotic species found, was more abundant in the reforested area, which may be due to its ability to colonize disturbed soils (Economidou and Yiannitsaros 1975). Any increase of habitat invasibility after plantation establishment is an undesirable restoration cost and may occur when the site preparation results in a high soil disturbance. A greater number of alien plant species was recorded in disturbed plots than in undisturbed ones in North American forests (Stapanian et al. 1998). To minimize this impact, reforestation projects could be designed using low site preparation methods (e.g. manual planting in pits).

In conclusion, it seems that the studied Mediterranean pine species, and especially *P. halepensis*, can be successfully used in reforestation projects of degraded Mediterranean kermes oak shrublands. However, whether these planting lead to the desired stable future ecosystem is a question that still remained unanswered. This should be validated by long-term studies, which should examine the ecosystem changes through several successional stages. Such long-term studies should be a priority in Mediterranean countries, where reforestation of degraded lands often comprises a basic tool to mitigate land degradation and also contribute to the maintenance of biodiversity (Vallejo et al. 1999).

References

Aronson J, Ovalle C, Avendano J (1993) Ecological and economic rehabilitation of degraded “Espinales” in the subhumid Mediterranean climate-region of central Chile. Landscape Urban Plan 24:15–21

Aronson J, Clewell AF, Blignaut JN, Milton SJ (2006) Ecological restoration: a new frontier for nature conservation and economics. J Nat Conserv 14:135–139

Atsamo A (2002) Early effects of four fast-growing tree species and their planting densisty on ground vegetation in Imperata grasslands. New For 23:1–17

Augusto L, Duponey J, Ricard J, Ranger J (2001) Potential contribution of the seed bank in coniferous plantations to the restoration of native deciduous forest vegetation. Acta Oecol 22:87–98

Baeza MJ, Valdecantos A, Alloza JA, Vallejo VR (2007) Human disturbance and environmental factors as drivers of long-term post-fire regeneration patterns in Mediterranean forests. J Veg Sci 18:243–252

Buford MA, Burkhart HE (1987) Genetic improvement effects on growth and yield of loblolly pine plantations. For Sci 33:707–724

Carnevale N, Montagnini F (2002) Facilitating regeneration of secondary forests with the use of mixed and pure plantations of indigenous tree species. For Ecol Manag 163:217–227

Dafis S (1973) Classification of forest vegetation of Greece. Sch Agric For Sci Yearb 15:75–90 (in Greek)

Economidou E, Yiannitsaros A (1975) Studies on the alien flora of Greece III. Morphology, development and phenology of Solanum elaeagnifolium. Cav Candollea 30:29–41

Fahnestock JT, Detling JK (1999) The influence of herbivory on plant cover and species composistion in the Pryor Mountain Wild horse range, USA. Plant Ecol 144:145–157

Ferris R, Peace AJ, Humphrey JW, Broome AC (2000) Relationship between vegetation, site type and stand structure in coniferous plantations in Britain. For Ecol Manag 136:35–51

Hatzistathis A, Dafis S (1989) Reforestation-forest nurseries. Giahoudis-Giapoulis, Thessaloniki (in Greek)

Hatzistathis A, Ganatsas P, Georgiou A (2000) Research on the improvement of reclamation techniques of mine land in Gerakini, North Greece. In: Tsihritzis VA, Korfiatis GP, Katsifarakis KL, Demetrakopoulos AC (eds) Proc. Int. Conf ‘Protection and restoration of the environment V’, Thassos, vol II, pp 699–706, 3–6 July 2000

Hobbs RJ, Norton DA (1996) Towards a conceptual framework for restoration ecology. Restor Ecol 7:93–110

Honnay O, Bossuyt B, Verheyen K, Butaye J et al (2002) Ecological perspectives for the restoration of plant communities in European temperate forests. Biodivers Conserv 11:213–242

Huston MA (1998) Biological diversity: the coexistence of species on changing landscapes. Cambridge University Press, Cambridge

Keenan R, Lamb D, Woldring O, Irvine T, Jensen R (1997) Restoration of plant biodiversity beneath tropical tree plantations in northern Australia. For Ecol Manag 99:117–131

Kerr G (1999) The use of silvicultural systems to enhance the biological diversity of plantation forests in Britain. Forestry 72:191–205

Lugo AE (1997) The apparent paradox of reestablishing species richness on degraded lands with tree monocultures. For Ecol Manag 99:9–19

Lunt ID (1990) Species-area curves and growth-form spectra for some herb-rich woodlands in western Victoria, Australia. Aust J Ecol 15:155–161

Maestre FT, Cortina J (2004) Are *Pinus halepensis* plantations useful as a restoration tool in semiarid Mediterranean areas? For Ecol Manag 198:303–317

Magurran AE (1988) Ecological diversity and its measurement. Univ. Press, Cambridge

Naveh Z (1990) Ancrent man’s impact on the Medlterranean landscape of Israel - ecological and evolutionary perspectives. In: Bottema S, EntJes-Nleborg G, Van Zetst W (eds) Man’s role in the shaping of the eastern Mediterranean landscape. Balkema, Rotterdam, pp 43–50

Norusis MJ (1994) SPSS professional statistics 6.1. SPSS Inc. SPSS Inc, Chicago

Onaindia M, Mitxelena A (2009) Potential use of pine plantations to restore native forests in a highly fragmented river basin. Ann For Sci 66:305–313

Perez-Trejo F (1994) Desertification and land degradation in the European Mediterranean. European Commission DG XII, Luxembourg

Pitkänen S (1997) Correlation between stand structure and ground vegetation: an analytical approach. Plant Ecol 131:109–126

Rebele F, Lehmann C (2002) Restoration of a landfill site in Berlin, Germany by spontaneous and directed succession. Restor Ecol 10:340–347

SER (2004) The SER primer on ecological restoration. Society for Ecological Restoration, Science and Policy Working Group. www.ser.org

Stapanian MA, Sundberg SD, Baumgardner GA, Liston A (1998) Alien plant species composition and associations with anthropogenic disturbance in North American forests. Plant Ecol 139:49–62

Thirgood JV (1981) Man and the Mediterranean forest. Academic Press, New York

Tsitsoni T, Ganatsas P, Zagas T, Hatzistathis A (1999) Rehabilitation of degraded forest ecosystems. AUTh, Thessaloniki, Greece, Technical Report, p 16

Tutin TG, Heywood VH, Burges NA, Moore DM et al (1993) Flora Europaea, 2nd edn, vol 1–5. Cambridge Univ Press, Cambridge

Vallejo VR, Bautista S, Cortina J (1999) Restoration for soil protection after disturbances. In: Trabaud L (ed) Life and environment in the Mediterranean, advances in ecological sciences, vol 3. WIT Press, Southampton, pp 301–343

Vidacovic M, Krstinic A, Durasovic P, Kajba D (1990) Thriving of some species and hybrids of two-needle pines in the Arboretum Trsteno. Annales pro Experiments Foresticis, vol 26. Zagreb in Jugoslavia MXM, pp 331–339

Yirdaw E (2001) Diversity of naturally-regenerated native woody species in forest plantations in the Ethiopian highlands. New For 22:159–177

Zagas T, Ganatsas P (2001) Results of a forest species trial in an Arboretum of North Greece. Silva Gandavensis 66:76–88

Zagas T, Hatzistathis A, Tsitsoni T, Ganatsas P (1998) Degradation of Mediterranean forest ecosystems and silvicultural measures for their restoration. European Environmental Research 1(2):53–60

Zagas T, Ganatsas P, Tsitsoni T, Tsakaldimi M (2004) Postfire regeneration of *Pinus halepensis* Mill. stands in Sithonia peninsula, northern Greece. Plant Ecol 171:91–99