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Pines and oaks in the restoration of Mediterranean landscapes of Spain: New perspectives for an old practice – a review

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Abstract

Pines have been extensively used for land restoration in the Mediterranean basin and in other parts of the world, since the late 19th century. The theoretical basis supporting pine utilisation was its stress-tolerant and pioneer features, and their attributed role of facilitating the development of late-successional hardwoods in the long-term. In the present work, the use of pines and hardwoods in forest restoration is discussed in the frame of the current disturbance regime and social demands for Mediterranean forests. Large pine plantations have recently disappeared because of their sensitivity to fire (e.g., *Pinus nigra*) or because of the short fire-intervals (e.g., *Pinus halepensis*). Combined pine and oak plantations are proposed for degraded land restoration on the basis of the complementary features of both groups of species. Seeding and containerised seedling plantation, soil amendments and plantation techniques to reduce transplant shock are evaluated for reforestation under water-stressing conditions, on the basis of several experiments performed in eastern Spain. Both *P. halepensis* and *Quercus ilex* are tested.

Introduction

Mediterranean pines have long been used for reforestation, especially since the 19th century. For instance, in the Algerian green belt, approx. 1 million hectares were planted starting in the 1970s (Lahouati 2000). Similarly, in Spain, 3.8 million hectares were reforested during the period 1945–1986, and 90% of the reforested area was planted with pines (ICONA 1989; Ortuño 1990). These massive plantations were carried out in all Mediterranean countries, mostly using pines, but also other conifers and eucalypts. For instance, the proportion of area reforested with conifers respect to total area reforested during the last decades was ca. 90, 94, 47, 55, 86 and 71% in Spain (1940–1984, Ortuño 1990), Turkey (1920–1997, Aslam pers. comm.), Algeria (Lahouati 2000), Morocco (FAO 2001), Portugal (1965–1995, Madeira pers. comm.), Greece (1941–2000, Direction of Reforestation and Mountain Hydrology 2001) and Tunisia (FAO 2001) (respectively). The objectives of these plantations were mainly to increase forest productivity, but also to protect watersheds and fix coastal dune systems. In addition these plantations contributed to provide employment in rural areas. Some of these old plantations on extremely degraded lands are today splendid pine forests.

The traditional strategy for reforesting degraded lands in the Mediterranean was first to introduce a fast-growing pioneer species, usually a pine species (Ceballos 1938; Gil and Prada 1993), assuming that this species would facilitate the introduction (either artificial or natural) of late-successional hardwoods (Barbéro et al. 1998). Nevertheless, this strategy was seldom completely applied because of the costly silvicultural post-plantation operations required and the current disturbance regime. In addition, due to the low dispersal ability of many Mediterranean hardwoods, and especially in the case of oaks (Vázquez 1998), spontaneous colonisation of late-successional species on pine plantations only occurred when seed sources

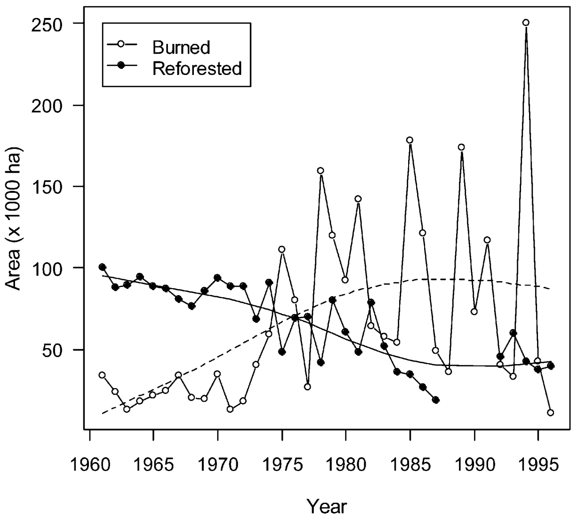

Figure 1. Annual wooded area burned (white symbol, dashed trend) and annual area reforested (black symbol, solid trend) in Spain during the last decades. Trends are significant ($P < 0.01$) smoothing lines. Elaborated from data in MAPA (1997).

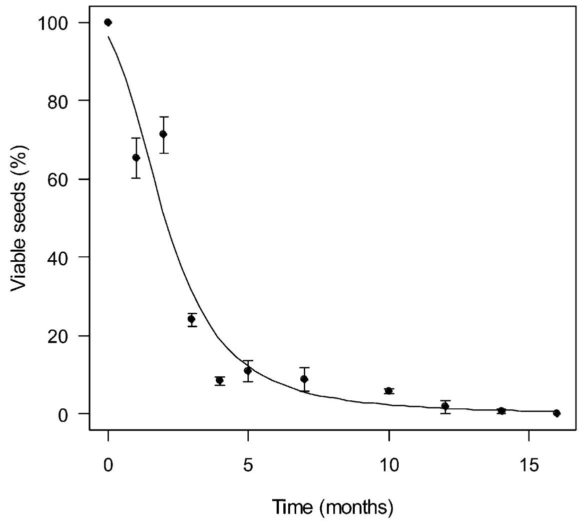

Figure 2. Proportion of remaining viable seeds of *Pinus halepensis* buried in the soil (mean and standard deviation of 4 sets with 40 seeds each). Results from an experiment carried out in a nursery near Valencia (eastern Iberian Peninsula) using soil from a recent burned area. Seeds were buried in December. Line corresponds to the logistic fit ($F_{1,9} = 115.25$, $P < 0.0001$) : $y = \exp(\text{lp}) / (1 + \exp(\text{lp}))$, where $\text{lp} = 3.217 - 2.9 * \ln(x + 1)$.

were very close by and the environmental conditions were favourable.

In the last decades, reforestation rates have been maintained and enhanced by the EU Common Agricultural Policy (CAP) in order to abandon marginal agricultural land. This policy tended to prioritise (by subsidising) hardwood plantations; for instance, in Spain from 1993–1996, the CAP policy was responsible for planting 238 112 ha, of which 44% were conifers and the rest hardwoods and mixed forests (Anonymous 1997).

In spite of the large reforestation efforts made in the various Mediterranean countries, the spread of large fires (Pausas and Vallejo 1999; Pausas in press) is compromising the persistence of these plantations. Extensive pine plantations resulted in large and homogeneous areas covered with flammable even-aged pines. These networks of pine plantations interconnected through old-fields, often colonised by flammable shrublands, have a high fire hazard and have facilitated the spread of the large fires occurring in the last decades. Changes in vegetation structure have probably contributed to the increase in the surface area burned in the last decades. On the other hand, reforestation rates may not be high enough to balance the surface area burned each year (Figure 1).

The effect of fire in pine woodlands

None of the pines in the Mediterranean basin are able to resprout, and thus their regeneration depends on their seedbank. For the most common lowland Mediterranean pines (*P. halepensis* Mill., *P. brutia* Ten. and *P. pinaster* Ait.), post-fire regeneration relies on the canopy seed bank protected in the serotinous cones, although the degree of serotiny for *P. pinaster* is very low (Tapias et al. 2001) compared with *P. halepensis* and *P. brutia*. Seed viability in the soil for these pines is short (Daskalakou and Thanos 1996). For instance, experimentally buried seeds of *P. halepensis* in the Valencia region showed a rapid decline in the number of viable seeds in the soil (Figure 2). The maximum decrease was observed between March and May, corresponding to the germination period, although seedling emergence (14.5 ± 8.3%) does not explain the strong reduction in the viable seedbank. One year after the onset of the experiment, no viable seeds remained in the soil (Figure 2).

In general, mature forest stands of these serotinous pine species regenerate after one single fire (Trabaud et al. 1985; Moravec 1990; Thanos et al. 1996; Ne'eman 1997; Herranz et al. 1997; Tsitsoni 1997; Arianoutsou and Ne'eman 2000; Leone et al. 2000). It has been suggested that fires may have fostered the spread of these species in the Mediterranean basin (Barbéro et al. 1998). However, as wildfires become more recurrent, the probability of affecting young pine woodlands (plantations or stands regenerating after a previous fire) is increasing. For instance, although both *P. halepensis* and *P. brutia* can flower at a relatively early age (<10 years-old, e.g., Thanos and Daskalakou 2000), they do not produce a significant canopy seedbank before the age of about 10–15 years (depending on environmental conditions), and thus, interfire periods shorter than 10–15 years may results in local extinction (Pausas 1999, Arianoutsou et al. 2002).

P. nigra Arnold and *P. sylvestris* L. do not have serotinous cones (Trabaud and Campant 1991; Tapias et al. 2001; Habrouk et al. 1999). In addition, their seeds are sensitive to the high temperatures produced during wildfires (Escudero et al. 1999; Habrouk et al. 1999; Nuñez and Calvo 2000). For these species, natural regeneration can only be expected in small patches preserved from fire, such as rocky outcrops, ridges, discontinuous vegetation mosaics (Escudero et al. 1999), and after low severity wildfires (Habrouk et al. 1999). For instance, large *P. nigra* mature forests have recently disappeared after a single fire in NE Spain, questioning the autosuccession capacity of these Mediterranean ecosystems (Rodrigo et al. 1999). *P. pinea* do not have serotinous cones either (Tapias et al. 2001), and some post-fire regeneration is due to the annual seed crop protected in closed cones, the high temperature resistance of the seeds conferred by the thick seed coat of this pine (Escudero et al. 1999), and the thick bark. This species mast seeds at 3–6 years interval, but some seeds are produced in the intervening years (Le Maître 1998). Other Mediterranean pine species such us *P. heldrechii* Christ and *P. leucodermis* Ant. do not survive fire (Blondel and Aronson 1999).

Apart from the effect of recurrent fires, under some circumstances one single fire can strongly modify the autoregeneration of serotinous pines. In a large survey of pine regeneration after fire in eastern Spain, we observed that *P. halepensis* regeneration was reduced at high altitudes (Pausas and Vallejo 1999), probably because plantations under these conditions are outside the optimum range for this species. Fire severity may also affect pine regeneration. Pine woodlands often show various degrees of canopy combustion mainly due to different fuel accumulation in the understory and to topographic features. This different degree of canopy combustion (fire severity) may determine the temperatures reached by the standing cones and, thus, the seed release and seed mortality. Whether the needles are combusted or not may also imply different post-fire soil litter inputs, which affect the soil conditions for post-fire germination. In eastern Spain, Pausas et al. (2002, 2003) found no differences in seedling densities between different fire severities during the first 29 months after fire; however, both seedling height and biomass were higher in sites affected severely than in sites affected with low severity. This pattern was related to higher post-fire soil P content under high severity events.

In central Spain, with a dry subhumid climate, Pérez (1997) observed almost no regeneration of *P. pinaster* after a single fire. She attributed this to competition with *Cytisus eriocarpus* Boiss. shrubland. Similarly, we observed a positive relation between pine seedling mortality and *Brachypodium retusum* (Pers.) Beauv. cover at small scale (Pausas et al. 2003). However, under drier (semi-arid) conditions with patchy vegetation, regeneration and growth of *P. halepensis* were higher in the vicinity of the stumps (Bautista and Vallejo 2002); the same occurred for grasses (mostly *Brachypodium retusum*). These results were related to the increased soil fertility close to the pine stumps. Similar results were obtained in pine woodlands of the Near East (Ne'eman et al. 1992; Ne'eman 2000). Therefore, under these conditions, fertility islands related to nurse trees improved seedling regeneration, even when the density of potential competitors, such as perennial grasses, was also high.

Despite the fact that forest surface area affected by fire exceeds the reforested area (Figure 1), if we compare two forest inventories (1967 and 1994; ICONA 1975; MAPA 1995) for eastern Spain (70% of the forest area dominated by *P. halepensis*), we observe an overall steady state or slight increase in forest land (Vallejo and Alloza 1998). This is due to natural regeneration, i.e., post-fire regeneration and old-fields colonisation. Thus, of the total forest area in 1994, ca. 20% was already forested in the first inventory, ca. 15% was planted, and ca. 65% corresponded to natural regeneration (figures refer to areas with more than 5% tree cover). These figures suggest that in eastern Spain, about 80% of these forests are less than 30 years old. Although natural regeneration together with plantations have maintained the surface area of forest land stable in the last decades, the increase in young forests and the increase in degradation drivers (e.g. climatic changes, forest fire risk, land abandonment and urbanisation; e.g. Piñol et al. 1998; Pausas in press) put this balance at risk. Thus, actions should be taken to prevent further reduction and degradation of forest land in Mediterranean landscapes. This degradation can only be reversed by human intervention in the form of restoration actions.

New perspectives

Although an increase in forest production is still an objective, current forest restoration actions in the Mediterranean area have new aims in order to comply with international obligations (e.g. Convention on Biodiversity, Convention to Combat Desertification, Convention on Climate Change): to increase carbon fixation, to increase biodiversity, and to reduce fire and erosion risk, as well as to increase rural development. As an example, the recently launched Spanish Forest Plan (2002) aims at planting ca. 125,000 ha annually, which is a prominent figure compared with previous reforestation programmes (Figure 1). In this context, reforestation techniques should adapt to increasing social demands for forests and wildland in general. In the specific case of forest restoration in fire-prone areas, where the main goal is nature conservation, the following major generic objectives are considered: 1) Soil and water conservation; 2) Improving the resistance, and especially the resilience of ecosystems with respect to wildfires; and, 3) Increasing mature woody formations, both forests and tall shrublands, depending on the environmental conditions of the site.

For sustainable reforestation actions in fire-prone Mediterranean landscapes, pines and broad-leaved resprouting species (especially oaks) should be combined to take advantage of the complementary features of both species groups (Table 1), i.e. the faster growth of pines (compared with Mediterranean oaks, Table 2; Castro-Díez et al. 1998) and the high fire resilience (efficient resprouting capacity) of oaks. The final aim is to increase the likelihood of plantation success and to reach the potential mature forest stage as soon as possible.

Early attempts to introduce broad-leaved resprouting species in the Mediterranean basin (e.g., *Quercus* species), faced high seedling mortality (Mesón and Montoya 1993), and until recently, nursery and field techniques for these species were poorly developed. Below, we briefly review some of the current restoration techniques for improving the restoration success of Mediterranean pines and especially of broad-leaved Mediterranean species. Examples are mainly taken from several experiments performed in the Valencia region (eastern Iberian Peninsula, Spain), an area with a typical Mediterranean climate and with vegetation dominated by *P. halepensis* woodlands. Potential vegetation of large parts of this area corresponds to *Q. ilex* L. ssp. *ballota* (Desf.) Samp. [= *Q. ilex* L. ssp. *rotundifolia* (Lam.) T. Moiras] forests (Bolòs 1967; Costa 1986), which currently occurs scattered over the landscape. Thus, in many of the experiments described below, both species are tested (and hereafter, the generic terms *Pinus* and *Quercus* refer to these species, otherwise stated). Experimental plots were installed on the two main bedrock types in the study area: limestones and mixed marl-limestone colluvium (called “marl" for simplicity). The soils developed on these substrates show contrasted properties. Soils developed on limestone, *terra rossa* type, are decarbonated brownish red *Rendzic Leptosols* (FAO 1988), shallow, discontinuous and very stony. Soils developed on marls are *calcaric regosols* and *calcaric cambisols* (FAO 1988), highly calcareous, relatively deep, and prone to surface crusting. All the experiments were performed on extremely degraded lands, with a long-term history of overexploitation and wildfires, on south facing mid slopes, and representing extreme dry conditions in the European context. The restoration techniques tested try to overcome water stress, which is the main cause of seedling mortality in dry Mediterranean conditions.

Seeding

When extensive young pine woodlands are affected by wildfires, natural regeneration is very difficult and slow. Therefore, artificial reintroduction of pines may be a suitable alternative to recover the forest in the medium term. Seeding is an attractive technique because of its low impact, low cost and easy application to remote and extensive areas (e.g. aerial seeding). Castells and Castelló (1996) obtained a relatively successful germination rate (ca. 5%) from an aerial seeding performed after a wildfire in Catalonia (NE Spain). In this study, seeds germinated in the first two months after seeding, coinciding with mild temperatures and abundant rainfall. Covering the seeds with a mulch layer could increase the germination rate and plant establishment (e.g., Muzzi et al. 1997; Brofas and Varelides 2000). For instance, in a post-fire experiment (eastern Spain), *P. halepensis* seeding plus mulching (straw) greatly increased the average seedling density (9.06 ± 4.48 individuals/m^2) com-

Table 1. Relative comparison of different traits between Mediterranean pines and Mediterranean oaks. Symbols + and − refer to higher and lower, respectively. Brackets are used when the attribute refers to some species only (i.e. only some pine species have persistent canopy seed bank).

	Pines	Oaks
Growth rate	+	−
Life span	−	+
Root:shoot ratio	−	+
Shade tolerance	−	+
Water-stress mechanisms	drought-avoiding	drought-tolerant
Dispersal capacity	+	−
Dispersal vector	wind	vertebrates
Desiccation tolerance	orthodox	recalcitrant
Seed bank	(persistent)	transient
Post-planting Seedling mortality	−	+
Resprouting capacity	none	+

pared to plots with spontaneous regeneration (0.47 ± 0.27 individuals/m^2). The application of a mulch layer alone (without seeds) slightly enhanced pine regeneration (1.33 ± 0.47 individuals/m^2), although there was a large spatial heterogeneity of germination rates. Furthermore, mulch can effectively contribute to reduce soil erosion in recently burned areas (Bautista et al. 1996; Badia and Martí 2000).

Seed predation is one of the major constraints for the direct seeding of pine species (Hadri 1975; Bergstern 1985). Predation of *P. halepensis* seeds applied by aerial seeding in late November 2000 was assessed in six sites affected by wildfires in eastern Spain. Most monitored *Pinus* seeds (>80%) were predated during the first six months, and at several sites, most predation occurred in less that one month (Figure 3). At the end of this experiment, germination was not observed in any plot, and so, restoration completely failed. In this study, predators were both birds and rodents, but they were not quantified; in spring (at the end of the experiment), ants become the most important predators (Acherar et al. 1984). Predation may be strongly site- and time-dependent, and thus highly variable. However, high predation risk represents a serious constraint for the direct use of this technique at a management level. So far, commercial repellents are not efficient enough to prevent predation (Llacuna 1998). Satiation of potential predators (e.g., by using large amounts of seeds of the target or complementary species) could be an alternative. On the other hand, even if predation rate is low, pines form a transient seed bank in the soil (Figure 2). Thus,

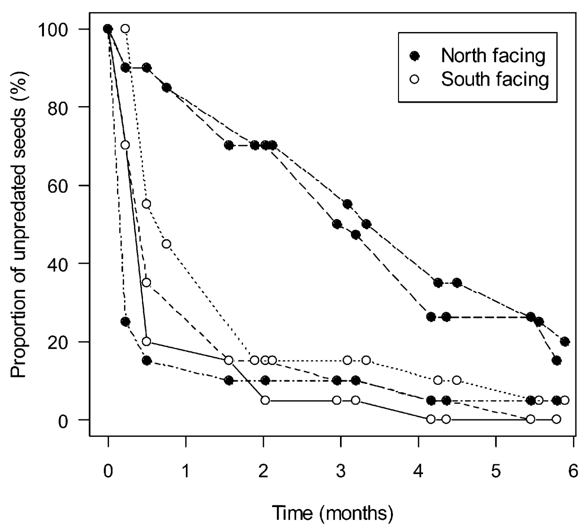

Figure 3. Proportion of non-predated *Pinus halepensis* seeds at 6 recently burned sites (different lines) with different facing slopes (different symbols) in the Valencia region (eastern Spain). One set of 20 seeds was sown on the soil surface in each of the six sites in January 2001 and monitored for 6 months (each seed was sown at least 15 m away from each other). The experiment was performed by the Forest Service of Valencia and it aimed to evaluate the effect of predation in burned areas that were recently seeded using aerial means.

the timing for seeding may be very relevant in order to take advantage of the most appropriate conditions and the relative short window for germination and establishment. The success of oak seeding is discussed below and compared with plantation techniques.

Planting

Due to the problems associated with seeding, the plantation of nursery grown seedlings is commonly preferred. Planted seedlings of *P. halepensis* have higher survival and early growth than most broad-leaved trees such as oaks (Figure 4). In eastern Spain, oak survival is frequently very low, especially when seedlings are planted on limestone (Figure 4) as was growth (data not shown). Under a semiarid climate (Vilagrosa et al. 2001), *P. halepensis* showed a low survival rate similar to some broad-leaved species like *Pistacia lentiscus* and *Rhamnus lycioides*, but growth was significantly higher (six years after outplanting). Differences in water-use efficiency associated with drought strategy are a key factor affecting plant survival and growth in Mediterranean conditions (Table 1). In fact, *P. halepensis* seedlings showed a significantly higher natural enrichment in ^{13}C ($\delta^{13}C$) than *Q. ilex* seedlings 20 months after outplanting under dry Mediterranean conditions (Valdecantos 2001), suggesting a higher water use efficiency of the former species. Water availability and plant water-use efficiency may be manipulated by different restoration techniques, such as: seedling preconditioning, water harvesting, soil amendments, and the use of treeshelters and nurse plants. Irrigation has not been considered so far in most of the current Spanish restoration practices at a management scale, although it is not unusual under semi-arid and arid conditions (Allen 1995; Lovich and Bainbridege 1999).

Preconditioning

In many cases, a key factor in plantation success is the transplant shock, that is, the initial short-term stress experienced by seedlings as they are transferred from favourable nursery conditions to the adverse field environment. Seedlings commonly change their morphology as they are transferred to the field (e.g. reducing the height:diameter ratio, Figure 5), suggesting that manipulating seedling morphology before outplanting may help to reduce the shock. Drought-preconditioning has been tested for various Mediterranean species (e.g. Nunes et al. 1989; Ksontini et al. 1998; Vilagrosa et al. 2003), and although it seems an attractive technique for Mediterranean conditions, it has shown poor to moderate results. For instance, Vilagrosa et al. (2003) found a clear positive effect of preconditioning on *Pistacia lentiscus*, but not on *Q. coccifera* and *Juniperus oxycedrus*. Villar-Salvador et al. (1999) reported few morphological and physiological modifications in *P. halepensis* seedlings subjected to short-term drought-preconditioning. Species response may be related to their drought strategy, and it may be necessary to design drought-preconditioning techniques according to this (Vilagrosa et al. 2003).

Table 2. Mortality and monthly relative growth rate in height (RGR) for seedlings planted in standard holes (control) and for seedlings planted with small runoff collection areas (water harvesting). Data correspond to 50 plants for each treatment, 16 months after planting, in Ayora (Valencia, eastern Iberian Peninsula, Spain). Means with different letters indicate significant differences ($p < 0.05$) between treatments.

	Control	Water harvesting
Quercus ilex		
Mortality (%)	49	30
Height RGR (month^{-1})	0.020 a	0.022 a
Pinus halepensis		
Mortality (%)	15	9
Height RGR (month^{-1})	0.0319 a	0.0360 b

Water harvesting

Runoff harvesting aims to intercept runoff and redirect water to the target seedling; it can be archived by sub soiling (deep regolith drilling) or by creating small runoff collection areas up-slope of the planting holes (microcatchments). This is a traditional technique for improving the water availability and site productivity in semi-arid and arid conditions (Whisenant, 1999; De Simón et al. 2001). For both *P. halepensis* and *Q. ilex*, seedling mortality (16 months after plantation) was reduced by the creation of small runoff collection areas in eastern Spain (Table 2), although growth only increased significantly for *Pinus*. Thus, this technique can be considered as a suitable way to increase plantation success under dry Mediterranean and arid climatic conditions.

Tree-shelters

Tree-shelters or protective tubes are used to modify the physical environment of the planted seedling ("mini-greenhouses"). If conveniently designed they can help to reduce evaporative demand and improve overall seedling performance (Bergez and Dupraz 1997; Bellot et al. 2002). In a *Q. ilex* experiment in the Valencia region, tree-shelters were used in three types of restor-

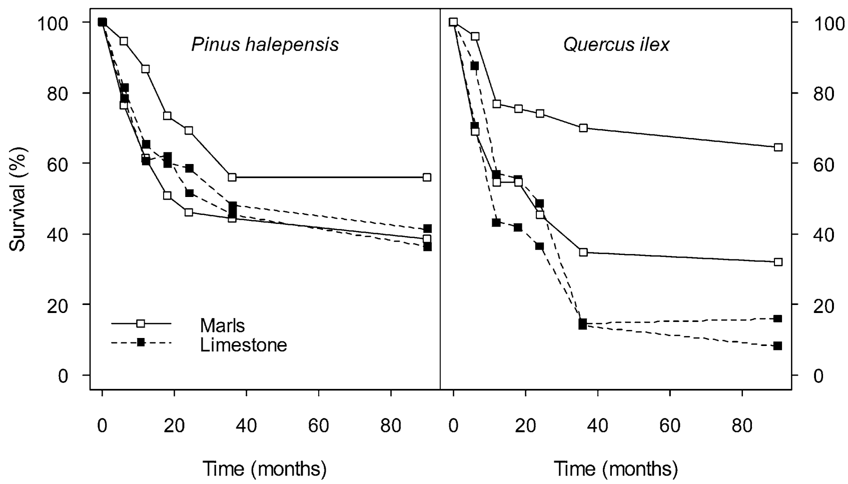

Figure 4. Survival of *Pinus halepensis* (left) and *Quercus ilex* ssp. *ballota* (right) seedlings planted under mesomediterranean conditions on marls (white symbols) and on limestone (black symbols) for 7.5 years (90 months) after outplanting. Data from 4 plots (in Valencia region, eastern Spain) where 75 *Quercus* and 75 *Pinus* seedlings were planted in an interspersed design.

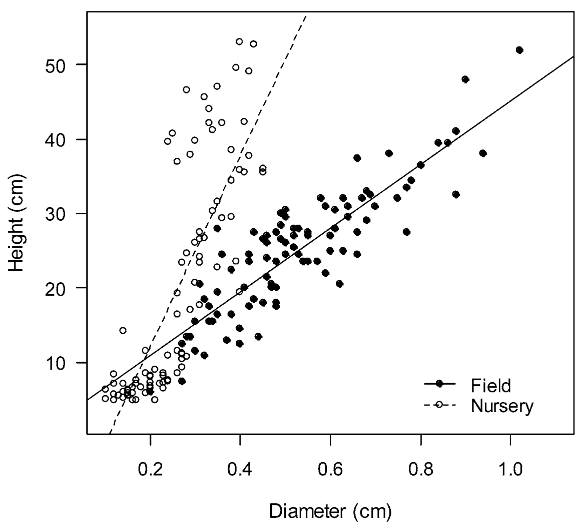

Figure 5. Height - diameter relationships for 1-year-old *Pinus halepensis* seedlings growing the nursery (open circles, dashed line) just before outplanting and for the same seedlings 18 months after planting in the field (i.e. seedlings of 2.5 years old) (black circles, solid line). Outplanting was performed between January and March 1993. Lines are linear fits (for nursery seedlings: Height $= -13.71 + 128 * \text{Diameter}$, $R^2 = 0.66$, $F_{1,97} = 194.4$, $P < 0.0001$; for seedlings 18 months after planting: Height $= 2.367 + 42.781 * \text{Diameter}$, $R^2 = 0.77$, $F_{1,93} = 303.1$, $P < 0.0001$).

ation actions: planting 1-year-old containerised seedlings, planting acorns pre-germinated in nursery, and directly sowing the acorns. Tree-shelters significantly increased the survival of pre-germinated acorns, but neither of 1-year-old seedlings nor the acorns showed increased survival (Figure 6). Tree-shelters reduced predation in pre-germinated acorns and thus improved their survival. Sown acorns showed predation and limited germination when unprotected by the tubes. Tree-shelters increased seedling growth for all tested techniques (Figure 7). Similar results were obtained in old-fields of SE Spain by Bellot et al. (2002). In another experiment under drier (semi-arid) conditions and shallow soils (Alacant, SE Spain), tree-shelters increased both survival and growth of *Q. coccifera* (Cortina et al. in press). Thus, tree-shelters can improve the survival and, especially, the growth of oak species under dry conditions.

Fertilisation and soil amendments

Mediterranean forest soils, especially degraded soils, are often deficient in phosphorus. Phosphorus limitations can be alleviated by fertilisation, which in turn may improve water use efficiency (Sheriff et al. 1986). Organic fertilisation can promote the growth of *P. halepensis* and *Q. ilex* seedlings (Valdecantos 2001; Figure 8). Application of organic amendments commonly results in lower root:shoot ratios, probably reflecting the reduction in limitation by belowground resources. Querejeta et al. (1998) also obtained improved growth and survival of *P. halepensis* in Murcia (SE Spain) with the addition of urban solid refuse, with secondary positive effects from deep soil preparation and mycorrhiza inoculation. Positive results of amendments with composted urban refuse have been obtained for *P. halepensis*, *Q. ilex* ssp. *ballota* and

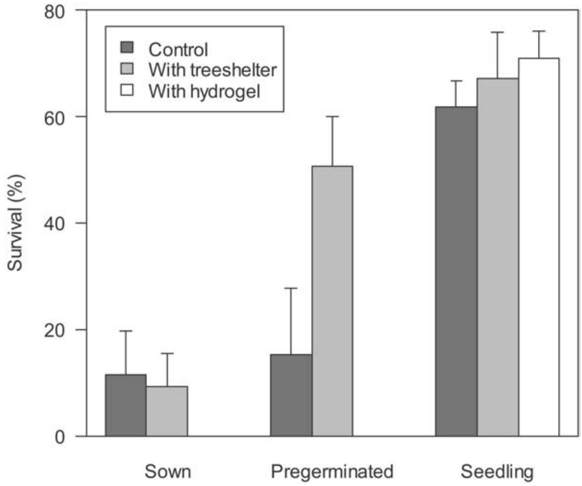

Figure 6. Survival of *Quercus ilex* ssp. *ballota* seeds and seedlings 4.5 years after plantation using three techniques: directly sown acorns, acorns pre-germinated 1 week prior to planting and standard 1-year-old seedlings, with and without tree-shelters. The initial number of seedlings/acorns was 50 per treatment. The experiment was carried out on southern slopes, on marls and under dry Mediterranean climate in Ayora (Valencia, eastern Spain). Values are mean and standard deviation of three sites. Tree shelters had a significant effect only on pre-germinated acorns.

Rhamnus lycioides (Valdecantos et al. 1996; Seva et al. 1996; Cortina et al. in press).

Hydrophilic gel polymers (hydrogels) are frequently used as soil amendments as they increase the soil water-holding capacity and may reduce the evaporation rates (Choudhary et al. 1995; Hüttermann et al. 1999; Tripepi et al. 1991). To what extent this additional water is enough to improve seedling performance in dry conditions is not yet clear. In eastern Spain, with medium textured soils, the application of dry hydrogel in the planting hole at a rate of 5 g per hole increased neither survival (Figure 6) nor growth (Figure 7) of *Quercus* seedlings. Callaghan et al. (1988, 1989) found that hydrogel amendments in sandy soils promoted seedling survival and growth under arid conditions, while Viero et al. (2000) under similar conditions found only an increase in seedling growth when hydrogel was applied in combination with watering. Contrasting results may be related to the soil texture; thus hydrogel application in sandy soils promotes an increase in water retention capacity and plant water potential (Hüttermann et al. 1999), while in loamy soils the effect may be negligible.

Biotic interactions (facilitation and perch effect)

Another way to increase water and nutrient availability for introduced seedlings is by recognising the spatial variability of these resources in the field. This involves selecting the microsite for planting close to a resource island or a nurse plant that would facilitate the survival and/or growth of the introduced seedling (Callaway 1995). The facilitating effect could be associated with shade, change in soil properties (e.g., increased infiltration) and retention of soil and nutrients provided by the nurse plant (Maestre et al. 2001; Castro et al. 2002), and also to the protection from grazers (Castro et al. 2002). This nurse effect has been observed in some Mediterranean species such as *Stipa tenacissima* with respect to *Pistacea lentiscus* (Maestre et al. 2003), *Salvia lavandulifolia* (Castro et al. 2002) and in some spiny shrubs (*Berberis vulgaris, Prunus spinosa* and *Rosa* spp.) conferring grazing protection to *P. sylvestris* and *P. nigra* (Castro et al. 2002). However, the nurse effect of *Pinus* has been questioned. Maestre et al. (2003) performed a manipulative experiment planting broad-leaved resprouting shrubs and trees in an open area (outside the pine canopy) and under the pine canopy. The results suggested that, under semi-arid conditions, the changes in understorey microclimate associated with *P. halepensis* were not sufficient to facilitate the establishment of broad-leaved resprouting shrubs; furthermore, survival was increased when grass (mostly *Brachypodium retusum*) was suppressed (Maestre pers. comm.), suggesting that the key factor for the negative net effect of pines was the competition with grasses. Grass-seedling (*Brachypodium-Pinus*) interference has also been suggested for recently burned pine woodlands of eastern Spain (Pausas et al. 2003). Therefore, considering previous results under other nurse plants, the balance between competition and facilitation is difficult to predict, and can go one direction or the other depending on the disturbance regime and resource availability (Bertness and Callaway 1994).

The role of plant-animal interactions has also been suggested for cost-efficient restoration plans (Handel 1997). In Mediterranean old-fields, secondary succession is faster in woody crops than in non-woody crops thanks to the role of trees as perch sites for frugivorous birds (the perch effect) (Bonet and Pausas in press). These birds eat and then defecate seeds of late-successional (bird-dispersed) species that germinate around the perching tree forming a nucleus of advanced succession (Verdú and García-Fayos 1996).

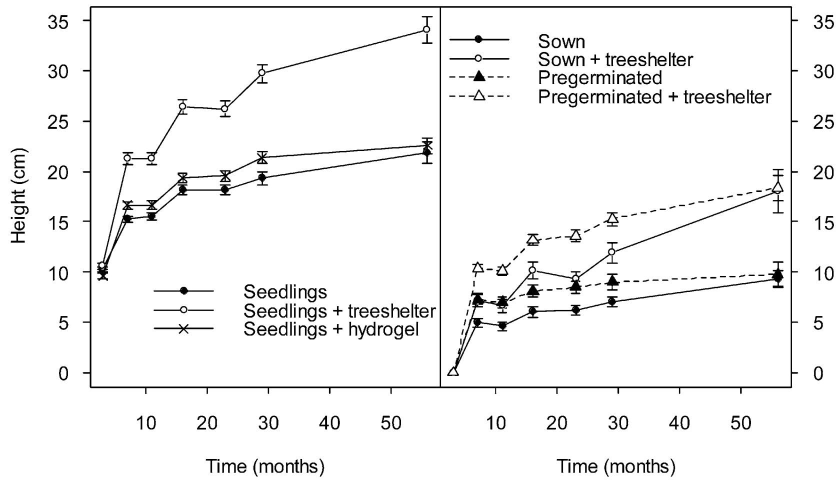

Figure 7. Effect of tree shelters and hydrophilic gels on the growth of *Quercus ilex* ssp. *ballota* seedlings (left) and effect of tree shelters on the growth of directly sown acorns and pre-germinated acorns of the same species (right). See experimental details in Figure 6. Values are means and standard errors for the surviving individuals of Figure 6.

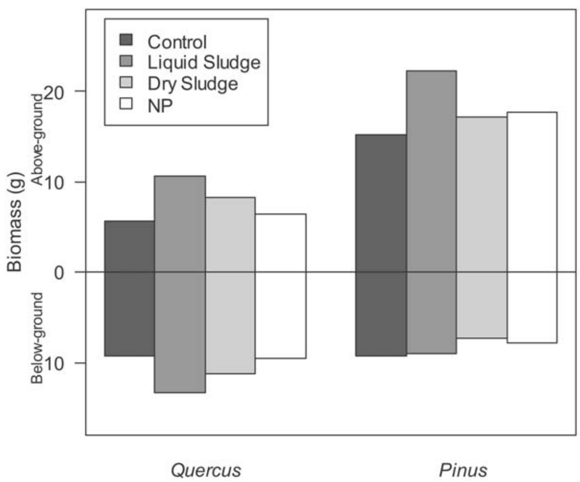

Figure 8. Biomass accumulation (dry weight) of *Quercus ilex* ssp. *ballota* and *Pinus halepensis* seedlings 20 months after planting on marls under dry Mediterranean climate in Ayora (Valencia region, eastern Spain), as affected by fertilisation: control (unfertilised), liquid sludge, dry sludge (both at an application rate of 10 mg dry weight ha^{-1}) and inorganic Nitrogen + Phosphorus (250 and 125 kg ha^{-1}, respectively). Values are means and standard deviation of 5 individuals. The decrease in the root:shoot ratio was significant for *P. halepensis* amended with liquid and dry sludge.

Furthermore, the perch tree may also create a favourable microsite (safe site) for germination and survival. This perch effect inspired a restoration technique based on providing bird perches (e.g., dead trees, artificial woody structures) in old-fields to accelerate colonisation rates (bird-mediated restoration). Despite of being an attractive technique to help succession, it has seldom been applied to Mediterranean ecosystems and most examples come from elsewhere (e.g., McClanahan and Wolfe 1993). A potential limitation for the use of this technique may be the lack of both suitable dispersal agents (Alcántara et al. 1997) and a nearby source of target species seeds; however, its application needs to be explored and tested under Mediterranean conditions. Furthermore, the strong relationship between oaks and its dispersal vector (Bossena 1979) merits a deep exploration in the light of reforestation of Mediterranean oaks.

Concluding remarks

Traditional strategies for the restoration of Mediterranean woodlands based on pines have sometimes failed mainly because of the failure to incorporate current disturbance regimes and the excessive simplification of predicting successional trajectories. In the light of modern landscape management, considering long-term effects and disturbance regime, and in the framework of the current social demands for forests, we suggest restoring Mediterranean landscapes using both pines and broad-leaved resprouting species, in order to take advantage of both the fast-growth features of pines and the high resilience of oaks. This will also provide higher diversity and landscape heterogeneity. However, the use of specific technology may be needed to improve water-use efficiency to increase the performance of these sensitive species. Specifically, in our study area we found that, under the current economic and technical constrains of extensive land restoration projects, seedling survival and growth can be increased by the use of water harvesting, tree-shelters, nurse plants and organic amendments. In many Mediterranean systems and due to large and longstanding human impacts, degradation processes are not local and extensive heterogeneous areas need to be restored. Different combinations of species (e.g. pines versus broad-leaved species) and different combinations of the above restoration techniques may be required for different purposes, and also for different parts of the landscape. Furthermore, some techniques seldom tested in Mediterranean conditions, such as the management of dispersal animals, may need to be considered in the future. Thus, landscape restoration programs should be diverse, adaptive, self-sustainable and they should take into account the ecological realities of change.

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