

Chapter 7

Post-Fire Management of Non-Serotinous Pine Forests

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7.1 Ecological Context

7.1.1 Short Definition/Justification of the Set of Forest Types Tackled in the Chapter

Nine *Pinus* species are found in the Mediterranean Basin (Barbéro et al. 1998). The respective forest types that are dominated by these pine species can be divided into four groups according to their ecological characteristics (EEA 2007):

- Thermophilous pine forests: Forests of *Pinus halepensis*, *P. pinaster*, *P. brutia* and *P. pinea* distributed at low elevations under thermo-Mediterranean climate.
- Black pine forests: Forests dominated by the various sub-species of *Pinus nigra* are regarded as typical of the mountainous zone of the Mediterranean Basin.
- Scots pine forests: *Pinus sylvestris* forests which are found across the higher, oro-Mediterranean altitudinal zone
- Alti-Mediterranean pine forests: Forests distributed near the timberline, with *Pinus heldreichii* and *P. peuce* as the dominant pine species.

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Although fire is generally recognized as an important ecological factor associated with pine-dominated forest ecosystems, not all pine forests share the same fire regime (Agee 1998). Pine species that have evolved under a fire regime characterized by frequent, high intensity events have serotinous cones in order to reassure their post-fire regeneration (see Chap. 6). This is the case of the thermophilous species *Pinus halepensis*, *P. pinaster* and *P. brutia*. The fourth thermophilous pine, *Pinus pinea*, does not have serotinous cones but it shows the highest resistance to fire in comparison to all the other Mediterranean pines due to the thickness of its bark (Fernandes et al. 2008).

All the other Mediterranean pine species are non-serotinous and can either persist fire if it is a surface, low intensity one or they are killed by extreme fire events and depend on seed dispersal from unburned sources in order to re-establish at the burned site (Arianoutsou et al. 2010a; Ordóñez et al. 2006). According to fire statistics, three out of the seven non-serotinous Mediterranean pine forest types (i.e. *Pinus pinea*, *P. nigra* and *P. sylvestris*) have suffered a considerable number of fire events during the last decades (Pausas et al. 2008; Rodrigo et al. 2007; Vilà et al. *in press*). This chapter is based on these three non-serotinous pine species.

7.1.2 Distribution of Non-Serotinous Pine Species

Pinus nigra (black pine) is as a typical Mediterranean pine species, with its natural geographic distribution ranging from Spain eastwards to Southern France, Italy and Austria, the Balkans and Turkey. It is also present in some localities of NW Africa, some of the Mediterranean islands (Corsica, Sicily, Cyprus and some Aegean islands) and the Crimean Peninsula in Black Sea (Fig. 7.1a). Its natural distribution also extends to southern Balkans and Anatolia. It is considered as a species capable to establish, grow and develop forest stands under a wide range of climatic and edaphic conditions, across an altitudinal zone that varies from 450 to 1,500 m.a.s.l (Dafis et al. 2001). For example, *P. nigra* is the only tree species of the meso-mountainous Mediterranean zone (800–1,600 m.a.s.l.) that forms forest ecosystems on ultramafic rocks. *Pinus nigra* forests are considered as priority habitats under the Annex I of the 92/43 Directive of the European Union. This is the result of two main factors: firstly, the high genetic diversity of *P. nigra*, accounting for the large number of sub-species across its natural geographical distribution, and secondly, its overall pattern of sporadic occurrence across the European Mediterranean countries (Fig. 7.1a).

Scots pine (*P. sylvestris*) is widely distributed (Fig. 7.1b). Its native range includes Scotland, Scandinavia (excluding Denmark), northern Europe, and northern Asia. Scots pine has some populations that are very well adapted to the Mediterranean mountain environment, particularly in NE Spain, S. France, N. Italy and the Balkans, with the Greek populations of Macedonia marking the southernmost limits of its natural distribution at the Balkan Peninsula (Barbéro et al. 1998; Dafis 2010). It prefers siliceous substrates, where it can form either pure forest stands or mixed stands with other high-altitude tree species, such as *Betula pendula*, *Picea abies*, *Fagus sylvatica* and *P. nigra*.

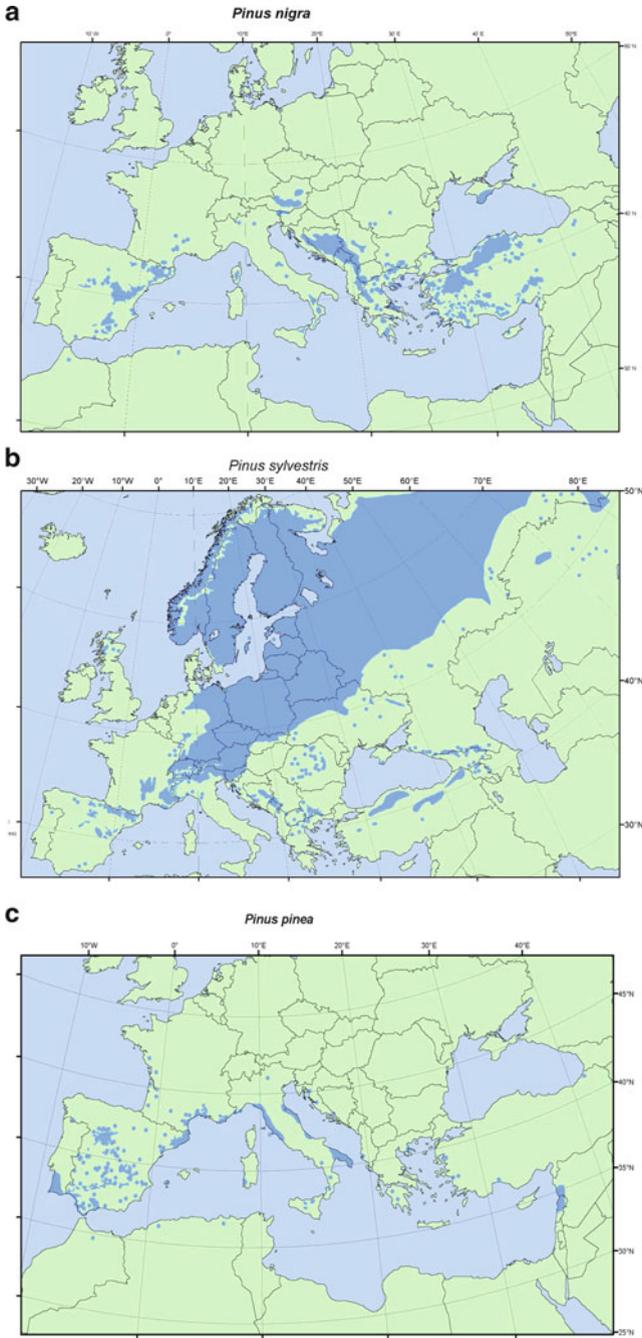


Fig. 7.1 Distribution of (a) *P. nigra*, (b) *P. sylvestris* and (c) *P. pinea* in the Mediterranean basin. Data obtained from Euforgen.org

Pinus pinea is a Mediterranean pine whose natural range is difficult to define because it has been planted in the Mediterranean Basin so widely for so long (Mirov 1967), but different authors suggest that it may have occurred naturally in the whole Mediterranean Basin (Le Maitre 1998; Martínez and Montero 2004). It occupies ca. 320,000 ha (Barbéro et al. 1998), 75% of them in Spain (Montoya 1990). Its distribution has been traditionally restricted to coastal sandstone and siliceous low elevation ranges (Fig. 7.1c). However as a consequence of the profitable historical exploitation of the pinion of this species for human consumption, its present distribution in the Mediterranean Basin is considerably wider than these sandy habitats. Now, in the Mediterranean Basin, *P. pinea* is common at low and intermediate altitudes and occurs in scattered populations (Barbéro et al. 1998; Dafis 2010).

7.1.3 Vegetation Composition in Non-Serotinous Pine Forests

The composition and structure of the major natural communities for the three non-serotinous pine species are significantly different among them (Gracia and Ordóñez 2011a, b, c). The three species form monospecific or mixed forests depending on climate, topography and disturbances. *Pinus pinea* does not usually form monospecific stands, in fact it is quite common to find this species mixed with other tree species such as *Quercus ilex*, *Quercus suber* or *Quercus humilis* and also some pine species such as *Pinus halepensis* or *Pinus pinaster*. *Pinus nigra* more frequently forms monospecific stands, but also shares the overstory with other species such as *P. halepensis*, *P. sylvestris*, *Q. ilex*, *Q. humilis* and *Fagus sylvatica*. *Pinus sylvestris* also forms monospecific forests or mixed forests with few species. In this later case, the most frequent tree species are *Q. ilex*, *Q. humilis*, *F. sylvatica*, *Pinus uncinata* and *P. nigra*.

The understory of the natural communities of the three pine species also varies depending on the pine species considered. Thus, in the western Mediterranean Basin, the most common species in *P. pinea* forests are *Cistus salviifolius*, *Cistus monspeliensis*, *Erica arborea*, *Rhamnus alaternus*, *Pistacea lentiscus* and *Arbutus unedo* (Gracia and Ordóñez 2011a). In the eastern Mediterranean Basin, the understory of *P. pinea* stands is occupied by typical species of evergreen sclerophyllous shrubs, species such as *Myrtus communis*, *Arbutus unedo*, *Erica arborea*, *Pistacia lentiscus* and *Phillyrea latifolia*, whereas whenever some gaps of the tree layer are encountered, seasonal dimorphic shrubs (primarily, *Cistus* spp.) are present (Dafis et al. 2001; Vassiliou 2007).

Pinus nigra forests in the western part of the Mediterranean have a diverse shrub and herbaceous layer dominated by *Buxus sempervirens*, *Juniperus communis*, *Crataegus monogyna*, *Prunus mahaleb*, *Thymus vulgaris* and *Viburnum lantana* (Gracia and Ordóñez 2011b). In the eastern part of the Mediterranean, a variety of factors, including climatic conditions, site pedology and topography and the stand history are expressed in the composition of the *P. nigra* forest understory. For example, at Mt. Olympus, the understory of *Pinus nigra* forests is dense,

occupied by evergreen sclerophyllous shrubs such as *Quercus coccifera* and *Arbutus* spp., whereas at higher altitudes the presence of the woody vegetation is reduced and the commonest species are *Rhus coriaria* and *Stachelina uniflosculosa* (Strid 1980). Species of the genera *Erica*, *Juniperus*, *Crataegus* and *Quercus* are common in the understorey of *P. nigra* throughout Greece (Dafis et al. 2001; Kazanis et al. 2011).

The vegetation associated to *P. sylvestris* forests from the Iberian Peninsula is dominated by *Buxus sempervirens* and *Juniperus communis*, accompanied by species such as *Vaccinium myrtillus* or *Arctostaphylos uva-ursi* in wet and mountain areas and by species of the genus *Quercus*, *Rhamnus* or *Erica* in drier ones (Gracia and Ordóñez 2011c). The commonest species in the understorey of *P. sylvestris* forests of Greece are *Pteridium aquilinum*, *Juniperus communis*, *Brachypodium sylvaticum*, *Rubus idaeus*, *Calamagrostis arundinacea*, *Fragaria vesca*, *Clinopodium vulgare* and *Rosa arvensis* (Dafis et al. 2001).

7.1.4 Ecological and Socioeconomic Importance

The ecological and socioeconomic importance of the different species of non-serotinous pines in the Mediterranean Basin is very high. The timber of *P. nigra* and *P. sylvestris* is of high technological quality, with both species having large and straight trunks. Their wood is used for many purposes such as construction, floor boards, saw logs, pulp and fuel. Additional products of these species are bark for gardening and resins. The timber of *P. pinea* is of mediocre quality and has short durability. The wood of this species is used as structural timber, sawn timber for light construction purposes, containers, as well as pulp for cellulose and paper (Giordano 1988). Although since the Roman period *P. pinea* has been artificially spread and cultivated also for timber, the pine nut of *P. pinea*, used in confectionery and food industries, has been an important commercial product from the ancient Egyptians and the Romans to present (Le Maitre 1998). However, it has been progressively replaced by pine nuts imported from other areas of the world, which involves low interest from private or public owners in carrying out plantations with this species in burned areas. Additionally, pinewoods of this species are able to provide other non-wood products such as resin, bark, honey and grazing for livestock.

Ecologically, *P. nigra* and *P. sylvestris* have been usually planted for erosion control and for reforesting burned sites, and they have been recommended for planting on strip-mined lands. Due to its tolerance to poor soil conditions, *P. sylvestris* is widely used in binding loose sands and in land reclamation programs in temperate zones. *Pinus pinea* tolerates sandy soils and, for this reason, has been traditionally used to consolidate coastal dunes and protect coastal agricultural crops. This species also has high interest in recreational forests due to its umbrella-like crown. Some varieties of these three non-serotinous pine species are frequently used in ornamental gardening.

7.1.5 Post-Fire Regeneration

In areas affected by understory fires, several adults of these three species (*P. nigra*, *P. sylvestris* and *P. pinea*) escape relatively unscathed even if these fires are intense, because their thick bark helps them to survive fire by insulating the cambium against lethal temperatures (Agee 1998; Fernandes et al. 2008). However, the survival of adults in these areas affected by understory fire does not promote an establishment of new seedlings of these species (Rodrigo et al. 2007 for *P. pinea*)

After crown fires, the natural regeneration of tree species lacking mechanisms to overcome the effects of fire (Tapias et al. 2004) is severely constrained (Retana et al. 2002; Rodrigo et al. 2004). Any of these three pines have serotinous cones (Escudero et al. 1997; Tapias et al. 2001, 2004), they open all the cones each year and they are not able to store seeds in closed cones, as *P. halepensis*, *P. brutia* or *P. pinaster* do (see Chap. 6). Moreover *P. nigra* and *P. sylvestris* show a similar seeding phenology, with cones maturing over 2 years and dispersal of seeds occurring in late winter to spring of the third year (Laguna 1993; Skordilis and Thanos 1997), just before their germination season. Then, most of the cones in *P. nigra* and *P. sylvestris* trees affected by summer wildfires (which are the most common ones in the Mediterranean region) are empty at that time and the seed canopy is exhausted. Seeds that germinate in late spring are burned as seedlings during summer fire (Retana et al. 2002) and the few seeds that remain in the soil over the ground are not able to resist the high temperatures of the fire (Habrouk et al. 1999). Thus recruitment into the burned area is difficult; in fact field data in different studies show an almost nil regeneration of *P. sylvestris* and *P. nigra* after fires (Espelta et al. 2002; Rodrigo et al. 2004; Vilà et al. in press). On a similar way, *P. pinea* cones ripen in spring over 3 years (Ganatsas et al. 2008; Tapias et al. 2001), while some authors consider that cone opening and seed release extend until autumn (Tapias et al. 2004). Although the cones are not serotinous and open at low temperatures (Tapias et al. 2001) some of these seeds can potentially survive summer wildfires, which may explain some field data documenting low but non-zero values of *P. pinea* establishment just after fires (Rodrigo et al. 2004, 2007). However mortality of these seedlings is high and does not allow a significant recruitment of *P. pinea* in burned areas (Rodrigo et al. 2007).

Moreover, when dispersed seeds arrive at the burned area, this does not necessarily imply a good seedling establishment. *Pinus nigra*, *P. sylvestris* and *P. pinea* seeds are consumed by mammals, birds and ants. These groups of animals, especially small mammals and birds, show a drastic decrease in their populations just after fire, but they recover few years later (Prodon et al. 1987; Torre and Díaz 2004; Brotons et al. 2005). As a consequence seed predation just after fire would be low. Moreover, in the case of *P. pinea* the harvesting of pine nuts for human consumption could limit seed availability in certain areas. Another limiting factor for recruitment is the competition between new seedlings and other vegetation regeneration in the burned area, especially for *P. sylvestris* and *P. pinea* that are more shade-intolerant than *P. nigra*. Therefore the seedlings established during the first years after fire, when

vegetation cover is low, are those that have higher probability of survival (Ordóñez and Retana 2004). In consequence the regeneration in burned areas of these three pines is concentrated in a limited spatio-temporal window, when the predation of their seeds and plant cover are still low. The high growth and survival rates of these seedlings of *P. nigra* and *P. sylvestris* established during the first years after fire allows the recovery of these species in a narrow area close to unburned trees. On its hand, *Pinus pinea* shows high mortality rates when shrubby vegetation recovers, further limiting the recovery of this species (Rodrigo et al. 2007). This is even more evident if we consider that these pines achieve reproductive maturity around 15–20 years for *P. nigra* (Tapias et al. 2004), 15 years for *P. sylvestris* (Vilà et al. *in press*) and 10–20 years for *P. pinea* (Tapias et al. 2004). This time lag and the short dispersal distance of this species limit their overall regeneration rate in large burned areas.

7.1.6 Post-Fire Dynamics of Animal and Plant Communities in Non-Serotinous Pine Stands

It has been traditionally accepted that Mediterranean plant communities have a high resilience to fire, that is, the composition and structure of burned communities is restored very quickly, and the burned ecosystem cannot be distinguished from the predisturbance state after a few decades (Trabaud and Lepart 1980; Thanos 1999). However, recent studies (Retana et al. 2002; Rodrigo et al. 2004) indicate that Mediterranean basin forest communities and their dominant tree or shrub species show different responses after large fires. Thus, forests of seeder species that produce few seedlings after fire and have limited long distance dispersal (as it is the case for *P. nigra*) are replaced by other vegetation types. As trees have a key effect on the composition of the communities where they grow (Ne'eman et al. 1995), the composition and structure of whole plant communities after fire are directly related to the regeneration of the dominant tree species in the canopy (Arnan et al. 2007). In this context, *P. nigra* forests showed low regeneration after fire (Arnan et al. 2007). The lack of the main tree species (Retana et al. 2002) agrees with the pattern shown by the whole plant community, with low similarity composition values between burned and unburned plots. Although there were not great differences in dominance or diversity in unburned and burned *P. nigra* forest plots, the fact that the canopy does not re-establish completely or recovers to another vegetation type (Retana et al. 2002; Rodrigo et al. 2004) determines changes in the plant species present and causes large variation between burned and unburned plots (Arnan et al. 2007; Arianoutsou et al. 2010b).

Recently, a database was formed, including species that are known to inhabit mature *P. nigra* forests in Spain and Greece in order to realize the degree of our knowledge regarding their response to fire (Kazanis et al. 2011). The database consists of 131 taxa (with the pine species included), 47 woody and 84 herbaceous. It was proved that for most plant taxa of the database there was adequate knowledge

supporting their ability to regenerate after a fire event. With the exception of ‘annuals’ and ‘sub-shrubs’, for all the other growth forms the commonest post-fire regeneration mode is resprouting. It should also be noted that it is among the ‘perennials’ that most species with remaining unknown response to fire are found. The percentage of taxa characterized by anemochorous and zoochorous seed dispersal is high. In the former group, taxa with small, light and numerous seeds are included, predominately of the Asteraceae and Poaceae families. In the later group, either taxa with fleshy fruits (endo-zoochorous dispersal, e.g. Rosaceae, Fagaceae) or taxa with fruits bearing spines (exo-zoochorous dispersal, e.g. Fabaceae, Poaceae) are included. Regarding early post-fire establishment, seeds of anemochorous are more prone to arrive at a burned stand from unburned sites, since for zoochorous taxa the improbable arrival of animals at the burned site would have been required. Nevertheless, for both categories the importance of unburned forest stands is once more highlighted.

Regarding the post-fire patterns of animal communities, they clearly parallel with those of overall plant community. For instance, in a study with ants conducted in the same vegetation types than those of Arnan et al. (2007), ant communities of *P. nigra* forests were the least resilient ones (Arnan et al. 2006), and this was attributed to the fact that *P. nigra* forests show little resilience to disturbance, and after fire they are replaced by completely different communities, either by coppices of resprouter species or by grasslands (Retana et al. 2002; Ordóñez and Retana 2004; Rodrigo et al. 2004). Other study found a persistent replacement of ant species in burned *P. nigra* forests (Rodrigo and Retana 2006), as it is also the case with vegetation. Such explanation is also applied to ground beetles communities, which in burned forests of *P. nigra* remain different from those in unburned areas along a long-term post-fire chronosequence (Rodrigo et al. 2008). In the case of birds, post-fire community is also strongly associated to vegetation recovery even when strong vegetation changes occurred due to non-direct regeneration of dominant forest tree species after a large fire. This is the case of *P. nigra* and *P. pinea* forests, where the non-direct regeneration process might create the appropriate habitats for open habitats species (Ukmar et al. 2007; Zozaya et al. 2011) and, thus, induce a turnover in species. As a deviation to this general trend, we are aware about a work with rodents where no differences in composition were detected between burned and unburned habitats of *P. nigra* few years after the fire (Ordóñez and Retana 2004).

7.2 Post-Fire Management: Issues and Alternatives

Ecological traits of non-serotinous pines considerably constrain the range of post-fire management options applicable in fire affected areas. The lack of seed and seedling survival of these species implies that their natural post-fire regeneration depends closely on the existence of *seed sources*, either isolated trees or groups of trees that survive fire both within the burned area and in the edges (Retana et al. 2002; Ordóñez and Retana 2004; Arianoutsou et al. 2010a). After the fire, survival and seed production of these trees determine the potential for colonization of the burned area.

Pinus pinea trees survive crown fire considerably better than other Mediterranean pines (Rodrigo et al. 2004), and their thick bark (Rigolot 2004; Mutke et al. 2005) make adult *P. pinea* trees relatively resistant to intense fire (Tapias et al. 2001). As a consequence, the presence of small groups of surviving trees of this species in areas affected by crown fire was relatively frequent. Rodrigo et al. (2007) found that 62.5% of *P. pinea* plots burned with crown fire in Catalonia had at least one *P. pinea* individual alive after fire. On the contrary *P. nigra* and *P. sylvestris* show lower survival of isolated trees in small groups of trees into the burned area (Rodrigo et al. 2004). But in the case of large fires often occur areas not affected by the fire, *green islands*, which can have different sizes and allow also the presence of groups of alive trees into the burned area. For *P. nigra*, Ordóñez et al. (2005) have found that trees, especially large trees, located in these small “green islands” produce more cones and more frequently than those in the edges or in large islands, and represent an important seed source for the post-fire regeneration of this species.

The success of post-fire regeneration from these seed sources depends, however, on the distance of seed dispersal. The distance at which pine seeds are dispersed is usually short. For *P. nigra* in a burned area in NE of Spain the distance observed was less than 50 m (Ordóñez et al. 2006), and reports of similar or even lower distances were obtained in Mt. Parnonas (S. Greece) (Arianoutsou et al. 2010b; Kakouros and Dafis 2009). For *P. sylvestris*, Vilà et al. (in press) have found that 90% of recruits were located at less than 25 m from the pines in the edge of the burned area. For *P. pinea* this dispersal distance is even shorter, with a maximum distance of 10–15 m from the crown (Montoya 1990; Rodrigo et al. 2007). The short dispersal in this species is related to seed morphology and weight: it is the heaviest seed of all Mediterranean pines, and has a rudimentary wing, considerably shorter than the seed itself (Tapias et al. 2001, 2004) which renders wind dispersal difficult. In areas affected by large fires where the proportion between burned area and unburned perimeter or green islands is very low and where most of the burned surface is far from seed sources, this type of regeneration is restricted to a little proportion of burned area or to small burned areas (Gracia et al. 2002). The competition of natural regeneration with resprouters and seeders (shrubs and oaks mainly) further limits possibilities of natural regeneration of these forest types after fires.

Under these conditions, successional trajectories after fire disturbance would most likely lead to the conversion of pre-fire non-serotinous pine dominated communities into other vegetation types (shrubland, grassland, oak-dominated woodlands), unless active restoration of pine woodland is implemented through reforestation measures (see Sect. 7.3). Post-fire management practices currently applied in non-serotinous pine forests reflect these issues. Table 7.1 reports the main practices applied in the management of non-serotinous pine forests affected by fire, as reported through a specific questionnaire by a sample of European countries in the framework of the FP0701 COST Action. Post-fire logging, mainly salvage logging, is widely and timely applied. The lack of any active post-fire response mechanisms explains well the widespread application of active restoration measures (planting or seeding) within 3 years after fires.

Table 7.1 Example of post-fire management measures applied in the case of non-serotinous pine forest (Processed from COST Action FP0701 data)

Post-fire management practices	Bulgaria	France	Greece	Italy	Latvia	Lithuania	Poland	Tunisia
Post-fire logging (Y/N)	Y	Y	Y	Y	Y	Y	Y	N
Post-fire logging timing (< 3 months after fire; 3–6 months; > 6 months)	3–6	3–6	3–6	3–6	< 3	< 3	< 3	–
Post-fire natural regeneration (Y/N)	Y	N	N	Y	Y	Y	N	Y
Active seeding or planting (Y/N)	Y	Y	Y	Y	Y	Y	Y	N
Active seeding/planting timing (<1 year after fire; 1–3 years; >3 years)	1–3	1–3	1–3	< 1	1–3	1–3	1–3	–

There is no paradigm or context to recommend active restoration to support the re-establishment of non-serotinous pines in fire affected areas. Alternative options exist and are mainly related to the conversion into more fire resilient communities, e.g. oak dominated woodlands (see Sect. 7.3.1). Reforestation option must be carefully evaluated being aware that restoration goals are often limited by the claims of other land uses, (see Sect. 7.3.1) and should be recognized as valuable by policy makers and public opinion (e.g., high conservation value, see Sect. 7.3.2). When planning restoration actions, land managers must be also aware that developing ecosystems, because of their low fire resilience, may undergo rapid transitions that would not result in the recovery of the components of the pre-fire forest community; thus, it is critically important to increase fire-resilience by enhancing the presence of resprouters, in order to promote self-regenerating forest types as independent as possible from further external subsidies. There are relevant experiences of post-fire management of Scots pine stands in Alpine environments: in such cases, the interventions are tuned to climatic conditions quite different from the Mediterranean ones (Beghin et al. 2010).

7.3 Case Studies

7.3.1 *Early Post-Fire Management in P. nigra Forests in Central Catalonia (Western Mediterranean Basin)*

7.3.1.1 The Wildfires

Since 1990, large wildfires have destroyed more than 25% of the total area occupied by *P. nigra* forests in Catalonia (NE Spain) (Gracia et al. 2000). Central Catalonia was affected by two of the largest historically recorded wildfires: the Bages-Berguedà

fire, which burned ca. 24,300 ha of forested land in July 1994, and the Solsonès fire, which burned ca. 14,300 ha in 1998. Prior to these fires, these areas had not burned for at least 70 years. The climate of the region is dry-subhumid Mediterranean (according to the Thornwaite index) with mean annual temperature between 10° and 13°, and annual precipitation between 550 and 750 mm. According to the data provided by the Ecological Forest Inventory of Catalonia (IEFC) carried out in 1993 (Gracia et al. 2000), natural *P. nigra* ssp. *salzmanii* forests were dominant before the fires occurred (78% of the burned surface), with *Q. ilex* and *Q. cerrioides* being extensively present in their understorey. *Pinus halepensis* forests were also represented in 14% of the burned area. Non-forested areas were mainly represented by croplands (Gracia et al. 2000). The resulting burned land was characterized by areas of high spatial heterogeneity in the distribution of burn severities. Thus, the fire left a mosaic of surviving green islands (10–15% of the total area affected by fire, Román-Cuesta 2002) immersed in a charred matrix.

7.3.1.2 Land Use Changes After Fire

An extensive survey of the entire burned area of the Bages-Berguedà wildfire, carried out through the comparison of aerial photographs prior (1993) and after (2005) the fire (Fig. 7.2), coupled with an extensive field sampling, revealed the existence of important land use changes (14% of the burned surface) (Espelta et al. 2002). Due to the regeneration failure of *P. nigra*, the area previously occupied by this species was extraordinarily reduced (from 15,700 ha to less than 100 ha). On the other hand, due to the vigorous resprouting of oaks, these species largely increased their presence in the landscape (from 1,048 ha to 12,450 ha). Transformation of previously forested areas to new croplands and rangelands accounted, respectively, for 1,005 (4.1%) and 2,365 ha (9.7%) of the 24,300 ha burned. However, a detailed analysis of land use changes points out that their distribution in the whole burned area is aggregated. Development of new croplands areas is positively related with the amount of previous croplands in the area and negatively related to the regeneration of resprouters. On the other hand, new rangeland areas are positively linked to the amount of surface where regeneration of resprouters is successful. These land use changes can be interpreted in the light of three major influences: (a) the need of landowners to find economic alternatives to forest logging, (b) the conviction of local authorities that favouring agricultural practices can decrease forest continuity and, thus, help to avoid the recurrence of large wildfires, and (c) the EU agricultural policy in the recent years, which has subsidized some crops and extensive livestock in the Mediterranean area.

7.3.1.3 Post-Fire Regeneration of *P. nigra* in the Study Area

Tree Survival and Cone Production

Tree survival and seed production were analyzed during 5 years as a function of tree size, crown damage, and tree location (i.e. location in edges or in islands)

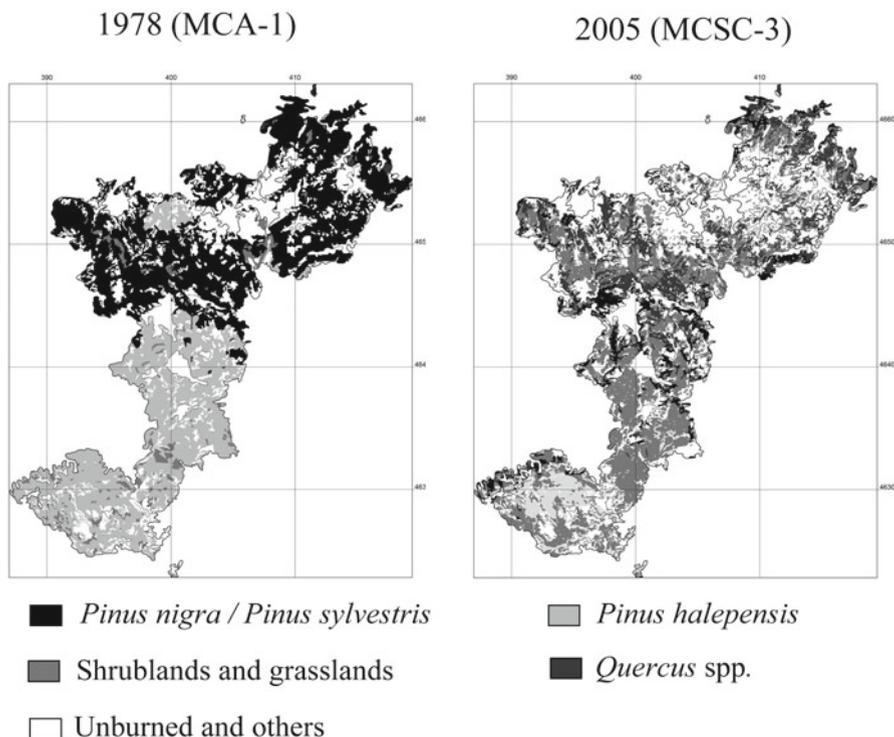


Fig. 7.2 Comparison of land covers in the area of the Bages and Berguedà regions affected by a large wildfire in 1994. *At left*, the map of land cover types in 1978 (Primer Mapa de cultivos y aprovechamientos, MCA-1). *At right*, the map of land cover types in 2005 (Tercer Mapa de Cobertes del Sòl de Catalunya, MCSC-3). Land cover types has been grouped in five categories. Source: Joanjo Ibañez, CREAM

(Ordóñez et al. 2005) in the area burned in 1998. Survival of *P. nigra* trees 5 years after fire increased with tree size (DBH > 20 cm) and decreased with crown damage. However, the response of trees with different fire damage varied with location, as less-affected trees showed higher survival in islands, while more affected ones performed better in the perimeter of the burned area. The main factor determining cone production during the first years after fire was tree size, because large trees produced more cones and more frequently than small ones. No differences were observed due to crown damage, but differences in cone production according to tree location were important. Thus, small trees produced cones more frequently in islands than on edges, while both cone production and the proportion of years that each tree produced cones decreased with island size. These results highlight the role of large trees and unburned islands as the main post-fire seed sources.

Seedling Establishment

The establishment of seedlings of *P. nigra* was examined under experimental controlled conditions of light and water as well as under natural conditions in the field (Ordóñez et al. 2004). The results suggest that seedling establishment after fire is scarce in field conditions and under a wide range of degrees of plant cover. However, *P. nigra* behaves as a species more shade tolerant than other pines but, given that this is not a common condition in recently burned areas, its regeneration is strongly influenced by fire.

The effects of seed predators (ants, rodents and birds) on post-dispersal seed removal and early seedling establishment of *P. nigra* were also evaluated by means of selective enclosure experiments limiting their access to seeds (Ordóñez and Retana 2004). The results indicated that the overall predation rates of *P. nigra* seeds by the three groups of predators were quite high, and the contribution of each group to overall predation showed seasonal variations. In the seedling establishment experiment, only in the exclusion treatment of the three predator groups there was initial establishment in all habitats, especially in the recently burned area. As in other species, seed predation can strongly limit population recruitment of *P. nigra* by reducing seed availability.

Post-Fire Regeneration Patterns

Post-fire regeneration patterns of *P. nigra* in the burned area of Bages and Berguedà were analyzed by developing a model of succession to predict medium-term changes in forest composition 30 years after fire (Rodrigo et al. 2004) from the regeneration monitored during the first years after fire (Retana et al. 2002). The results show that although *P. nigra* was the dominant species in the area before the fire, it almost disappears after the fire because its seedling density is almost nil. The highest proportion (76.7%) of plots originally dominated by pines changes after fire to communities dominated by oaks (*Q. ilex*, *Q. cerrioides*). Although a high proportion of the pre-fire oak seedlings and saplings present in the understory of *P. nigra* stands dies after the passage of fire, there is still a considerable percentage that sprouts vigorously and allows oak dominance in the future forest. There is also a considerable percentage of burned pine plots that change to shrublands or grasslands of *Brachypodium* spp. The transformation of these pine forests into shrublands or oak woodlands seems related to the length of time since abandonment, with the more recently abandoned lands becoming shrublands and the older areas, those providing time and a suitable habitat for oaks to establish (Lookingbill and Zavala 2000), becoming oak woodlands. Moreover, 7% of plots dominated by *P. nigra* before the fire showed a large post-fire regeneration of the serotinous pine *P. halepensis*, originating from seeds produced by the rare *P. halepensis* trees present in the canopy before the fire. Transformation into *P. halepensis* forests would be related to the proximity to seed sources of this pine or the fact that *P. nigra* can also form mixed forests with *P. halepensis*.

Difficulties and Predictions for the Future

Three processes linked with the beginning of the recruitment, that is, 1) seed survival after fire, 2) seed predation, and 3) seedling survival, do not predict good perspectives for *P. nigra* forests after large wildfires. Thus, most seeds are dispersed in late winter and have already germinated in spring (Habrouk et al. 1999). Consequently, fire burns these seedlings, while the few seeds remaining in the soil are unable to withstand the high temperatures attained during intense summer wildfires (Habrouk et al. 1999). Moreover, post-dispersal seed predation by different animal groups is also high, as they consume many of the *P. nigra* seeds remaining on the ground. Finally, this low seed availability is particularly important taking into account that seedling survival is also very low during the first year after germination (Franco 2001). All these results suggest that the natural post-fire recovery of *P. nigra* in burned areas is difficult. Thus, natural regeneration of non-fire-prone seeder species after fire relies entirely on the arrival of propagules from seed sources, such as isolated trees present within the burned area or those at the unburned edges (Espelta et al. 2002). However, the distance at which pine seeds are dispersed is usually short, less than ca 50 m for *P. nigra* (Ordóñez et al. 2006), and larger trees (the main seed sources) are scarce in the study area. Under these constraints, colonization from the surrounding unburned landscape is likely to take many decades.

Ordóñez et al. (2006) developed a simulation model for predicting the recruitment response of *P. nigra* from unburned edges in the study area. The distribution of *P. nigra* seedlings at different distances from the unburned edges was simulated by integrating empirical field data for the different processes affecting seed (cone production, pre-dispersal cone predation, seed production per cone, seed dispersal, post-dispersal seed predation and seed germination) and seedling success. The model was successfully validated in old-burned areas. The simulated values of established seedlings 30 years after fire followed a normal distribution in the first 100 m, with a wide range of 2,000–25,000 seedlings/ha. The maximum dispersal distance showed a shorter range around 120 m. Thus, fires create large post-fire heterogeneity in seedling densities that depend on the distance to the seed source, but also on the characteristics of the forest before the fire. Plots with medium and large trees showed an increment in increasing seedling establishment with tree density, whereas plots dominated by small trees had very low regeneration. If we consider that *P. nigra* trees in the study area are concentrated in the smaller size classes, and that the forest front advanced in a much closed form, the natural post-fire recolonization of *P. nigra* in burned areas is very difficult.

Post-Fire Management Practices

Under these circumstances a major interest grew up in the last decades about the best alternatives to restore *P. nigra* forests throughout suitable artificial reforestation programs, and to ameliorate the structure of mixed oak coppices (Espelta 1999).

An extensive experiment combining different methods of vegetation clearing (mechanical, controlled burning or grazing), soil preparation (ripping or planting holes) and reforestation methods (broadcast seeding, spot seeding and planting) was conducted in view of assessing the best alternatives to restore *P. nigra* forests (Espelta et al. 2003). These practices were compared in terms of seedling establishment. The results showed that the final establishment of *P. nigra* seedlings 2 years after the experiment onset ranged drastically from 7 ± 4 seedlings/ha in the broadcast seeding treatment to 610 ± 40 seedlings/ha in the plantations. The failure of the broadcast seeding assay points out that, although it may be recommended to wait a precautionary time to observe whether natural regeneration occurs after a wildfire (Espelta 1999), the fast recovery of ground vegetation, as well as that of some animal groups (especially seed-harvesting ants), severely threatens the success of broadcast seeding treatments. Seedling establishment after sowing was very poor and not influenced by vegetation clearing. In plantations, seedling survival was higher in the ripper treatment than in planting holes for all vegetation clearing treatments except the control one. Although the establishment of seedlings of *P. nigra* obtained in the plantation experiences is low in comparison to success reported in temperate and boreal forests, it is in the range of other experiences carried out in Mediterranean environments (Vallejo and Alloza 1998).

7.3.2 Early Post-Fire Management in *P. nigra* Forests in Southern Greece (Eastern Mediterranean Basin)

7.3.2.1 The Wildfire

Mt. Parnonas (1,935 m) lies across the south-eastern part of the Peloponnese District, Southern Greece. It is characterized by a variety of habitats and a rich plant and animal diversity, which justifies its inclusion in the Natura 2000 network (GR2520006). An extended zone of coniferous forests, formed by *P. nigra* and *Abies cephalonica*, lies between 700 m and 1,700 m. For both species, the populations of Mt. Parnonas (together with those of the neighboring Mt. Taygetos) are on the southern-most edges of their natural geographical distribution. On the 23rd of August 2007, 1,921 ha of *P. nigra* forest were burned, i.e. about 36% of the total *P. nigra* forest cover on Mt. Parnonas.

7.3.2.2 Management Objectives

Taking into consideration the fact that *Pinus nigra* is not adapted to high intensity fire events and the importance of this specific pine population of Mt. Parnonas, a pilot action plan for the post-fire restoration of the burned forest was proposed and funded under the LIFE+ European Union Programme, titled “Restoration of *Pinus nigra* forests on Mt. Parnonas (GR2520006) through a structured approach”.

The current LIFE+ programme started in January 2009 and is expected to be completed in June 2013. Still, some preliminary results have been presented for the various scheduled actions. The actions scheduled in the context of the project were as follows (Kakouros 2009, <http://www.parnonaslife.gr/en>):

- Impact assessment of 2007 fire
- Demonstration of a structured approach for the restoration of *P. nigra* forests
- Implementation of restoration measures
- Monitoring and evaluation of the restoration

The impact assessment of fire was based on the detailed mapping of the burned areas and the evaluation of the fire impacts through the use of remote sensing, geographical information systems and fieldwork. The assessment showed that across the burned area, several forest stands remained unburned, accounting for a total area of 420 ha (Kakouros et al. 2009). As mentioned earlier, the importance of such stands acting as seed sources is very high.

The program involved the development and demonstration of a structured approach for the restoration of *P. nigra* forests that will help to define priorities for the restoration of the affected areas. Of major importance is the description of a step-by-step process for prioritizing and selecting the most suitable areas for restoration. Prioritization and selection is achieved by applying exclusion criteria, eligibility criteria in terms of abiotic parameters and technical criteria (Kakouros and Dafis 2010). In the case of Mt. Parnonas, the exclusion criteria were (i) the potential of natural regeneration and (ii) the potential of low survival of planted individuals. Following the first exclusion criterion all sites where pine seedling density exceeded or was expected to exceed 1 ind/m² were excluded. This was the case of sites near unburned stands or where fire burned only the understory. The second exclusion criterion corresponds to sites found outside the altitudinal limits that are regarded as the best for *P. nigra* growth, thus reducing the success of artificial reforestation.

The areas remaining after the application of the exclusion criteria have been ranked according to (1) the representativeness (sensu Annex I of the Habitats Directive) of the habitat type (i.e. areas with high representativeness before fire should have higher priority for restoration), (2) the inclusion of sites under conservation status (e.g. Natura 2000 sites or other protected areas), (3) the presence of important species (burned sites where rare or endemic species are known to be present should have higher priority for restoration), (4) the re-establishing forest connectivity (priority should be given to the restoration of forest stands that promote connectivity), and (5) the abiotic variables of the prospective areas (mainly soil depth and aspect), in order to select those with the higher potential of reforestation success. The final step is the consideration of the available resources (financial, personnel, seed stock) and the cost per hectare and per restoration method (seeding or planting) to determine the total area that will be restored.

After taking into consideration all the above mentioned criteria, the pilot restoration of 290 ha of *P. nigra* forests consisted of planting 464,000 *P. nigra* seedlings at 19 forest stands (Simadi 2010). The seedlings have been produced from seeds collected from cones of the Mt. Parnonas pine population in 2007 and reforestation begun in December 2010.

A crucial element of a restoration project, especially in cases where sites of the Natura 2000 network are involved, is the installation of a monitoring system for the evaluation of the restoration effectiveness. On Mt. Parnonas 33 permanent plots have been established, 13 plots for the monitoring of natural regeneration and 20 plots for the monitoring of artificial restoration (Kakouros and Dafis 2009). Valuable data are expected to be produced from the frequent sampling of these plots.

7.4 Key Messages

- Non-serotinous pine species do not regenerate after fire because there are no seeds available.
- The lack of active regeneration makes post-fire natural recovery mainly dependent upon seed dispersal from unburned patches.
- Natural succession trajectories after fire would most likely lead to the conversion of pre-fire non-serotinous pine forests into other vegetation types.
- This explains the widespread application of active restoration measures (planting or seeding) during the first years after fire.

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