Chapter 6
Post-Fire Management of Serotinous Pine Forests

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6.1 Ecological Context

6.1.1 Serotinous Pines

Fire response of pines was categorized by Keeley and Zedler (1998) according to two different strategies that are generally considered alternatives or complementary: individual survival (*fire resistant species*) and stand resilience (*fire evader species*). The first strategy is basically characterized by thick bark, long needles, thick protected buds, self-pruning, deep rooting, rapid growth and, in a limited number of species, resprouting capability. The second is characterized by the presence of a large canopy seed bank that ensures abundant post-fire seedling recruitment (*serotiny*); seeds are stored in the fruit or cones after maturation and the seed release occurs in response to an environmental trigger, fire in this case.

Fire resilient pines, characterized by the production of a high number of serotinous cones, small seeds and early reproduction capability, include a Mediterranean group of species primarily formed by *Pinus halepensis* Mill. (Aleppo pine), *P. brutia* Ten.

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(Brutia pine) and *P. pinaster* Aiton. (Maritime pine). According to the EEA (2006) forest type classification, they are all included in the category of the coniferous forests of the Mediterranean, Anatolian and Macaronesian regions.

Pinus halepensis and *P. brutia* are the main pine tree species in the Mediterranean Basin, especially in low altitudinal ranges, and both have high fire resilience (Thanos and Daskalakou 2000). *P. pinaster* has been formerly considered a non-serotinous species (Keeley and Zedler 1998) due to the polymorphism of this trait among and within populations (Tapias et al. 2001, 2004) but there are several populations showing a high degree of serotiny (Vega et al. 2010) since this species has been considered in this chapter. However, as this chapter is focused on serotinous pine species distributed in fire-prone habitats, other species such as *P. cembra, P. uncinata* and *P. mugo,* are not included. *P. radiata* is an introduced species for production purposes (plantations) and is not included as well.

6.1.2 Species Distribution

Serotinous pine trees are important components of many landscapes in the Mediterranean Basin and played a major role in the origin of the flora and vegetation (Barbero et al. 1998). The most common pine species, *Pinus halepensis* and *P. brutia*, are widely distributed around the Mediterranean Basin (Quezel 2000). Both species show taxonomical proximity and could be difficult to distinguish them out of their natural habitats where is frequent their hybridization. Together with *P. pinaster* Ait. which mainly grows in western Mediterranean Basin (Carrion et al. 2000), they are the three major conifers of the Mediterranean zone in terms of ecological and economic value and total area covered (Fig. 6.1).

P. halepensis cover is about 2.5 million ha mainly in western Mediterranean Basin (Iberian Peninsula, France and Italy) although it also occurs in northern Africa (mainly Morocco, Algerian and Tunisia) and locally in some locations of the eastern area (Egypt, Greece, Macedonia, Turkey, Syria, Jordan and Israel). It usually grows

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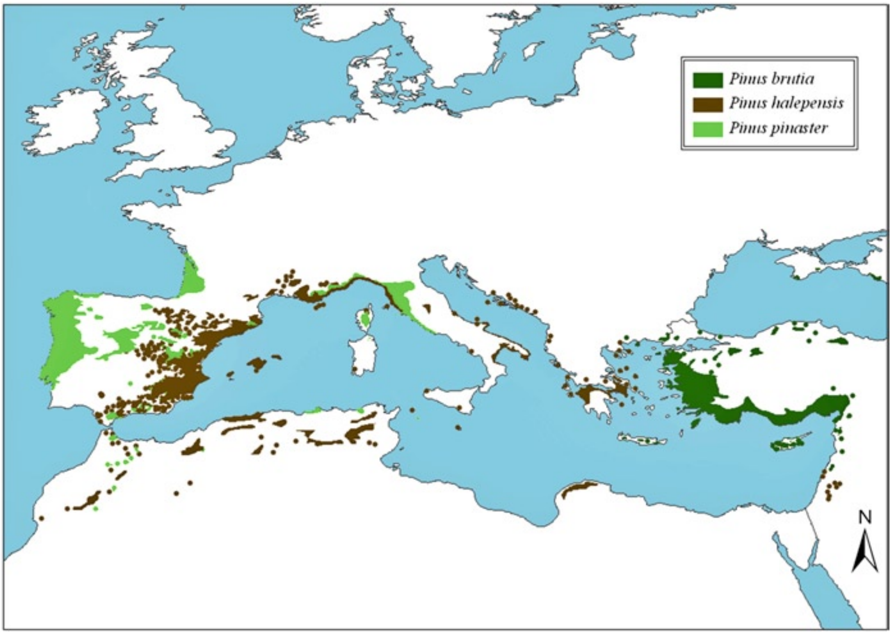

Fig. 6.1 Distribution of *P halepensis*, *P brutia* and *P pinaster* in the Mediterranean Basin (Source: Euforgen.org)

on marly limestones and marls, from the sea level up to high altitudes (more than 2000 m asl), although it prefers lower altitudes occurring mostly in the Thermo and Meso-Mediterranean. Its habitat ranges from the lower arid or semiarid to humid bioclimates.

P. brutia covers more than four million ha in the eastern Mediterranean Basin (Greece, Turkey, Kurdistan, Afghanistan, Iran, Syria, Jordan and Israel). Its cover has been increased with extensive afforestations in the nineteenth century, in France, Italy and Morocco. Similarly to *P. halepensis*, it grows on marly limestones and fissured soils with high altitudinal range (up to 2000 m amsl) although its optimal is in the Thermo and Meso-Mediterranean. This is stricter species in terms of water requirements and it is not frequent in arid or semiarid bioclimates.

P. pinaster covers more than 1.5 million hectares in western Mediterranean Basin, being found in the Iberian Peninsula, France, Italy and northern Africa (mainly in Morocco and Tunisia). Although it prefers sandy and acid soils, it also grows in basic soils and even in sandy and poor soils. The altitudinal range includes from the sea level to high altitudes (about 1500 m amsl) up to the Supra-Mediterranean belt. It resists frost but not hard droughts, consequently it does not occur in arid or semiarid bioclimates.

Due to the climatic conditions and the fire dynamics of Mediterranean Basin, the these *Pinus* species use to regenerate after fire and they shows early cone production

(Tapias et al. 2004), the total recovery and maturity (structurally and functionally), is estimated to require at least 20–30 years which has been known as immaturity risk (Trabaud 2000; Tsitsoni et al. 2004; Vega et al. 2010). Forest management should be prioritized depending on the vulnerability of the burned site. It should be characterized taken into account vegetation characteristics before the fire, climate conditions, soils and landform characteristics. The rate of vegetation recovery is the combined result of plant reproductive strategy (seeder or resprouter species) plus physical factors, such as climate and aspect. Recruitment of Mediterranean serotinous pines could be stimulated by fire (Boydak 2006; Pausas 1999a; Vega et al. 2010), but in the middle term, ecosystem vulnerability will depend on the ability to persist without major changes in vegetation composition, structure and the relative cover and biomass of the species present. In this sense, ecosystems dominated by obligate seeder species have lower resilience in the young phase due to the immaturity risk (Keeley et al. 1999).

6.1.3 Composition of Natural Communities

The three serotinous pine species form natural stands in the previously mentioned geographical areas although they have been introduced by reforestation in almost all the Mediterranean Basin (within and outside their natural range).

They form monospecific or mixed forests, depending on the soil, altitude and bioclimate. These forests can be an intermediate step in the succesional series to broadleaved trees, such as some species of *Quercus.* However, in some areas they conform climacic communities as paraclimaxes (Barbero et al. 1998), composed of *Pinus* or other coniferous species, such as *Abies pinsapo* Boiss., *Tetraclinis articulata* (Vahl) Mast, *Juniperus thurifera* L., *J. phoenicea* L. and *J. oxycedrus* Sibth. & Sm.

The natural communities of the three Mediterranean pines include different shrub species depending on soil, bioclimates and forest degradation but the most common species which can also be found in shrubland formations (*maquis* and *garrigue*) are *Rhamnus lycioides* L., *Quercus coccifera* L. or *Q. calliprinos* Webb, and several species of the genus *Pistacia, Cistus*, *Phillyrea*, *Genista*, *Anthyllis*, *Rosmarinus, Myrtus, Erica, Arbutus* or *Calluna.*

6.1.4 Ecological and Socioeconomic Importance

The three pine species are very important due to their ecological role and economical uses in the Mediterranean Basin (Alia and Martin 2003; Fady et al. 2003). Aleppo and Brutia pines represent the only or main source of wood and forest cover in many Mediterranean countries, such as *P. brutia* in Turkey or *P. halepensis* in Tunisia. Their wood is used for many purposes such as construction, industry, carpentry, firewood and pulp. Even seeds are an important resource for pastry in several areas, mainly in North Africa. The importance of *P. pinaster* comes mainly from the uses related to wood and the production of high quality resin. It can be considered a fast-growing species in the Atlantic region, where it is used for pulp and paper production, construction, chipboards, floor boards and palettes.

Ecologically, *P. pinaster* has been considered in its habitat region, the most important pine species for making recreational forests with high soil protection. On the other hand, Aleppo pine is the most important forest species in North Africa, and it has high ecological importance in southern France and Italy, especially at the urban-forest interface. The importance of *P. brutia* is similar in eastern Mediterranean Basin. In addition, the adaptation of the species to drought, poor soils and forest fires are the reasons by which they have been used in several afforestation programmes throughout the Mediterranean Basin, both for wood production and soil protection (Alia and Martin 2003; Fady et al. 2003).

6.1.5 Expansion or Regression of Populations

An increasing number of pollen fossils and charcoal records from different areas in the Mediterranean Basin have related the presence and/or expansion of the three pine species to fire events (Vega et al. 2010; Quezel 2000), although high fire frequency would have limited the presence of *P. pinaster* and *P. halepensis* (Christakopoulos et al. 2007; Gil-Romera 2010).

Current trends indicate that in North Africa, their distribution area decreased in the last decades due to human activity (land use change, overgrazing and overexploitation) whereas in northern part of the Basin, their geographical extent is increasing due to the expansion after fire and European Union policies promoting afforestation (Barbero et al. 1998). Global warming can be a factor to be considered in the next years in relation to the dynamics of the distribution area of the serotinous pine species although this has not been studied.

6.1.6 Fire Adaptions and Regeneration Strategies

The landscapes in the Mediterranean Basin have been transformed in association with the effects of fire, being the first evidence of fires induced by human activity from about 10,000 years ago (Boydak et al. 2006). In this fire-prone habitat, as it has been above mentioned, one strategy has been developed by some Mediterranean pine species, which are obligate seeders: the seed storage on the canopy protected in the pine cones to take advantage of the conditions that taking into account after fire, delaying the opening of cones to release seeds after the fire (Saracino et al. 1997; Tsitsoni 1997).

Serotiny is a term pertaining to the retention of mature seeds in a canopy-stored seed bank for 1–30 years or even more (Lamont et al. 1991). According to Richardson

(1998), serotiny is analytically defined as a morphological feature of some pine cones (and reproductive structures in other plants) whereby the cones remain closed on the tree for one or more years after seed maturation; cones open rapidly when high temperatures melt the resin that seals the cone scales. Long-term seed storage is common in dominant plant families of fire-prone areas, e.g. *Proteaceae*, *Cupressaceae* and *Pinaceae* (Tapias et al. 2001). The degree of serotiny in pines varies considerably among species and populations of the same species, mainly in response to fire frequency as well as site productivity (Ne'eman et al. 2004). Mediterranean serotinous pine species have been developed a dual strategy: post-fire obligate seeder (from serotinous cones) and an early colonizer (from non-serotinous cones) which attributes them high fire resistance. These pines regenerate easily after fires which trigger a massive release of seeds, but this also occurs in the absence of fires under suitable stand conditions (Nathan et al. 1999; Thanos 2000). Although the mature trees of the three considered pine species are tolerant to fire, life-history attributes such as a short juvenile period, large cone crops, fast growth and a massive regeneration capability after fire contribute considerably to fire resilience in these pine species (Thanos and Daskalakou 2000). Tapias et al. (2004) recorded a negative correlation between bark thickness and serotiny level in Iberian *P. pinaster* populations and linked the presence of serotinous cones in *P. pinaster* and *P. halepensis* to populations growing in areas with abundant scrub layer that are prone to very intense fires which frequently scorch or torch the crowns. In addition, they have developed the capacity for serotiny and are well adapted to fire prone habitats, mainly at low elevation along the Mediterranean Basin (Thanos and Daskalakou 2000).

However, serotiny is more than the delayed opening of pine cones since increase the probability of success to the recruitment (Moya et al. 2008c). The seeds released from serotinous cones reach better conditions for survival and germination (low competition) than those from non-serotinous cones and avoid excess predation (Saracino et al. 2004). In addition, serotiny produces a higher number of seeds that display the best biological qualities to be released after a forest fire (Daskalakou and Thanos 1996, 2004; Ferrandis et al. 2001; Leone et al. 1999; Saracino et al. 1997, 2004). These qualities include a higher number of sound seeds, greater weight and germination rate (Saracino and Leone 1994) or higher heat insulation and resistance [AU1] to unfavorable environmental conditions (Salvatore et al. 2010). In this way, serotiny in pine species maximizes the number of seeds available for the next generation by storing seed crops successfully and protecting them to ensure extensive and vigorous natural regeneration after fire or other disturbances.

Post-fire regeneration of *P. halepensis* has been extensively reviewed for the eastern and western Mediterranean Basin, with emphasis on community recovery, pine population reconstitution (Trabaud 2000), plant adaptive traits, pine forest communities and vegetation structure (Arianoutsou and Ne'eman 2000). The literature concludes that whole pine tree populations are usually burned by wildfires in *P. halepensis* or in *P. brutia* forests (Arianoutsou and Ne'eman 2000; Thanos and Doussi 2000). Taking into consideration that the three Mediterranean serotinous pines (*P. halepensis*, *P. brutia* and *P. pinaster*) are obligate seeders (Thanos and Daskalakou 2000; Tapias et al. 2001; Boydak 2006), the most critical point for natural regeneration and forest re-growth are the immaturity risk period and the amount of seeds stored in the canopy seed bank and the quantity of sound seeds available after fires (Thanos and Doussi 2000). Therefore, post fire regeneration of these species is strongly related to the quality and quantity of the canopy seed bank, since soil seed banks have a transient character (Ferrandis et al. 1996, 2001; Boydak 2006). In addition, early flowering and cone production is a common characteristic for both *P. halepensis* and *P. brutia* species with a short juvenile period, which probably ends at the age of 3–6 years for several pine individuals or at 12–20 years for the whole population (Thanos and Daskalakou 2000), although the latter species seems to start flowering and produce cones a little later (Boydak 2006). In mature stands, serotinous cones remain closed for several years, thus forming the canopy seed bank that is responsible for post-fire regeneration. Additionally, in young reproductive Aleppo pine trees, a high percentage of serotinous cones has been observed, in which resources are allocated to seed production and thereby reduce the 'immaturity risk' in the case of a second successive fire (Ne'eman et al. 2004).

Focusing on post-fire regeneration dynamics, massive seedling recruitment after fire has been extensively recorded (Boydak 2006; Daskalakou and Thanos 2004; Thanos and Marcou 1991; Tsitsoni et al. 2004). In many study cases, Aleppo pine seed germination, seedling emergence and establishment took place as a *single massive wave* shortly after the onset of the first post-fire rainy season which appears to be strongly related to rainfall and temperatures (Daskalakou and Thanos 2004), with no additional pine seedlings observed emerging during subsequent years. For *P. brutia* forests, densities of 10,000–30,000 biologically independent seedlings per hectare are proposed to be considered as successfully recruitment for natural regeneration after fire, whereas 3,000–10,000 seedlings are acceptable in certain parent materials and poor soils (Boydak 2006). On the other hand, mean post-fire seedling density (Thasos Island) was considerably high (20,000–60,000 seedlings/ha) at the end of the recruiting period. A significant drop in density was observed during the first summer and stabilization at 0.6–2.0 seedlings/m^2 was achieved 5 years after fire (Spanos et al. 2000).

Several studies reported the initial sapling density in the short term (3 years after the fire) and the survival rate, such as Brutia pine in Samos Island where the overall pine density was 0.15 saplings/m^2 6 years after a fire (Thanos and Marcou 1991), while values of 10,600–12,000 stems/ha at different sites were recorded in an artificial forest 20 years after fire (Tsitsoni et al. 2004). Moreover, sapling survival was found to be high (43%) in the sixth growing season after fire and was expressed by a negative exponential curve (Thanos and Marcou 1991). Sapling survival kinetics was described by a rectangular hyperbola, reaching a percentage of 28% 5 years after a fire (Spanos et al. 2000). For *P. pinaster,* an initial seedling density of about 8–12 seedlings/m^2 were found in Spain (Vega et al. 2010) and high variablitiy for *P. halepensis* was found in eastern Spain, from about 10,000 to 60,000 seedlings/ha, depending on latitude (Moya et al. 2008a). However, a high recurrence of fire has a strong, negative effect on the ecosystems, increasing the risk of degradation and soil loss. The minimum time interval to avoid this negative effects has been estimated in 15 years (Eugenio and Lloret 2004; Eugenio et al. 2006).

6.2 Post-Fire Management in Serotinous Pine Forests

6.2.1 Current Practices and Management Objectives

In general, the three serotinous pine species have been under two contrasting management alternatives: under-or over-managed.

Excessive harvesting and unplanned exploitation, both past and present, have been destroying the original forest structure to promote monospecific stands where pine growth is maximized by decreasing interspecific competition with other trees (oaks or other coniferous species). Typical treatments include clear-cut operations applied over large areas and the establishment of monoculture afforestation. This management approach has been developed under the premise of maximizing the potential timberline in short rotation ages. Although this could be a correct management in high quality site areas, it is totally wrong in low productivity regions (Bravo et al. 2008a, b) and the result, in several cases, are highly degraded forests resulting from [AU2] long periods of mismanagement (Çolak and Rotherham 2007).

In the last decades of twentieth century, the decrease of economic benefits from low productive areas for timber production lead to under-management, resulting in dense forests with almost null productivity rates and high fire hazard. The more common management practices after forest fires in the serotinous pine forests of Southern Europe are the salvage logging of the burned trees, usually carried out at early stages of succession (before 6 months after the fire). If the amount of seedlings is not enough to assure the natural regeneration, reforestation is usually carried out using the same main tree species. Stands naturally regenerated are not early managed and dense stands with lower productivity rates are developed.

In many cases, the current socio-economic and ecologic context in serotinous pine forests call for a more sensitive and sustainable forest management. External benefits such as recreational uses, decrease of rural depopulation, soil protection, biodiversity, carbon sink capability, reducing fire risk should be taken into account. There is an opportunity to introduce the current scientific knowledge to promote good practices.

The management of burned pine forests should first establish the main objectives of actions. In most cases, the prioritized objectives should include soil protection and water regulation, the reduction of disturbance (fire) hazard as well as increased ecosystem and landscape resistance and resilience to disturbances (fires), and the promotion of mature, diverse, and productive forests (Valdecantos et al. 2009). Foresters have to define whether they want to recover the pre-disturbance plant component of the ecosystem (i.e. a pure *Pinus halepensis* forest) or better moving the system towards alternative states (i.e. mixed pine-oak forest) following the classic restoration model proposed by Bradshaw and Chadwick (1980). The introduction of a larger and more diverse set of species in the restoration of pine forests with low regeneration may improve ecosystem resilience in the framework of current fire regimes (Valdecantos et al. 2009). However, both objectives may share common management tools and actions. An intermediate management goal must address avoiding future disturbances, such as fire, that convert the system to more degraded states. Selective clearing of obligate seeder shrubby vegetation has been shown to be an effective technique to transform a highly flammable, dense and continuous shrubland with a great amount of dead fuel, to grassland with sparse resprouting shrubs and regenerating pines and a discontinuous fuel load (Baeza et al. 2008). Mulching of the soil surface with brush-chipping of material from clearing greatly reduces the germination rates of obligate seeders (Tsitsoni 1997). In addition, the removal of standing vegetation by selective clearing may promote growth of remaining individuals and seedlings introduced by reforestation (Valdecantos et al. 2009).

6.2.2 Management of Burned Trees and Soil Protection

The first priority should be an emergency assessment of the burned area (Robichaud et al. 2009) (see Chap. 1). A multidisciplinary point of view is mandatory to the assessment of urgent threats and risks to human life, property and critical natural and cultural resources considered in an integrated plan. Following the procedures of BAER, Burned Area Emergency Responses (Napper 2006), the main topics to take into account should include forest fire severity, presence of water-repellent soils, mapping of effective soil cover, flood or debris risky areas, riparian stability assessment, potential erosion or sedimentation and water quality deterioration, and status of infrastructures. Immediately after forest fire soil erosion prevention may have to be carried out. When there is a low recovery rate of vegetation and/or where erosion is expected to occur, several management strategies might be applied. The application of an organic layer of mulch, either alone or combined with seeding native grasses, is a suitable rehabilitation option on steep slopes with low plant cover and high erosion risk, as it is aimed at reducing surface flow, rainsplash, soil crusting and compaction, thereby increasing infiltration. To be effective, this technique should be applied soon after the fire and before the heavy autumn rains, which means that areas vulnerable to erosion must be identified as soon as possible. In burned pine forests on steep slopes and soft soils, log dams or contour-felled barriers are also effective post-fire management practices for reducing physical soil degradation and erosion (Valdecantos et al. 2009).

In Aleppo pine stands, salvage logging has been checked to reduce the seedling density due to mechanical actions but also it induced the reduction of growth and the survival of pine seedlings due to the protection losses (Martínez-Sanchez et al. 1995). In addition, the burned wood promoted seedling recruitment due to microclimatic conditions which could be significant in poor site quality areas (Castro et al. 2010). These results are expanded and analyzed deeply in the study cases, Sect. 6.3.2.3.

6.2.3 Plantation and Seeding

Historically, extensive reforestation programs have been developed over the last two centuries in southern Europe, and more intensively in the last 50 years, resulting in expansion of the habitat distribution of these pines (Le Maitre 1998) due to the high availability of seeds, ease of seedling cultivation and success of survival in low fertility soils. In several countries, the reforested area is considerably larger than the original range. These plantations were undertaken for timber production and soil protection on degraded areas and wood production is currently an important economic resource on a local level.

In many reforestations, seed origin was not considered and this has become not only the main difficulty for clarifying migration pathways of the species, but more importantly, it may have drastically affected the genetic structure of populations with previous fire adaptive traits (Vega et al. 2010; Tapias et al. 2004), leading to difficulties in post-fire regeneration.

6.2.4 Management of Natural Regeneration

After the emergency treatments, the best option to support the restoration of serotinous pine forests is monitoring the burned area in order to record how natural regeneration progresses. After 1–3 years, there a new evaluation should be carried out and a management plan can be implemented (Corona et al. 1998). Depending on the success of natural regeneration, the options are: no action, assisted natural regeneration or active restoration. Forest restoration treatments often include reforestation with planting or seeding (less frequent) to complete the recovery in the burned area. When fire recurrence is lower than the required period for the community to reach maturity and to increase the resilience to new fires, the young stands could be burnt before the production of mature seeds. In this situation, artificial regeneration (reforestation) should be carried out with seedlings coming from similar provenances than the burned area to reinsert populations with similar resilience to the previous one. The success could be improved by adding sewage sludge (and, if necessary, pine seedlings inoculated with mycorrhizas), mainly in arid and semiarid areas (Fuentes et al. 2007; Gonzalez-Ochoa et al. 2003).

Regeneration capacity of pine forests after a fire will determine the most appropriate post-fire management option. As mentioned above, usually serotinous pine forests show high recovery after a fire, showing a moderate to high resilience. In fact, pine seed germination after fire is often excessive, promoting high intra-specific competition and low tree growth rates and, as a consequence, high fuel load accumulation. Post-fire management should consider all of these points, including fire recurrence. Simulation models of vegetation dynamics in relation to fire frequency show that *Pinus* species dominate the plant community at medium fire recurrence, declining at intervals longer than 100 years (Pausas 1999b). However, when these forests are affected by recurrent fires, succession can be diverted. A single fire is enough to change serotinous pine forests into alternative stable shrubland, where the colonization of late-successional species is impeded (Rodrigo et al. 2004). Undoubtedly, the management options should consider the forest conversion after the fire approaching the ecological window to reintroduce the climax species.

Furthermore, post-fire recovery may not be homogeneous within the whole burned area of a serotinous pine forest (Pausas et al. 2004b). For instance, the fire severity will determine the regeneration depending on the availability of seeds and scorched needles protecting the soil surface (Pausas et al. 2003). Furthermore, plant cover and type influence soil protection by decreasing hydrophobicity and soil losses (Cerda and Doerr 2005). Land use history and soil type, properties that are often intimately related, also determine the type and characteristics of the regenerating community after a fire. Baeza et al. (2007) observed that the regeneration of a mixed forest of *Pinus halepensis* and *P. pinaster* 23 years after fire was mainly driven by the occurrence of a second wildfire, land use and soil type, releasing six different vegetation types from grasslands to dense shrubland with young regenerating pines. As a consequence, the landscape and the plant community are quite heterogeneous just after post-fire regeneration, suggesting that specific local management schemes are needed. In burned Aleppo pine forests in eastern Spain, sapling recruitment has been observed to be positively related to the presence of branches and trunks covering the soil surface, which create a favorable microclimate for germination and seedling establishment (Pausas et al. 2004). [AU3] At a later stage, planting seedlings is suggested for increasing site diversity or reintroducing species that have disappeared after the fire.

In some cases, high-density stands are opened by fires or silvicultural treatments which reduces understory fuel accumulations limiting crown fire propagation (Fernandes and Rigolot 2007) and increase seed production (Moya et al. 2008a). Dense stands which receive little treatment have high potential for destructive crown fires (Pausas et al. 2008), although *P. pinaster* stands are generally vulnerable to fire in any conditions (Fernandes and Rigolot 2007). Fuel treatments by means of prescribed fire in the understory or alternative techniques can make these stands less vulnerable to crown fire. The interaction between provenance, fire severity and post-fire management can be critical for seedling recruitment success. Some of these aspects are specifically addressed in other sections of this chapter.

Although *P. pinaster* re-establishment after a single fire is generally acceptable in part of its natural area, there are post-fire regeneration difficulties in some of these populations. In burned areas where *P.pinaster* and *P.halepensis* populations are in contact, *P. halepensis* can be more effective for seed dispersal than maritime pine due to its larger seed bank, since it is more thermophyllous and tolerant to summer drought, and usually has a more abundant regeneration than maritime pine. More xeric conditions and more severe and frequent forest fires are expected, according to the climate change scenarios for the Mediterranean Basin over the coming decades (Moriondo et al. 2006). Consequently, it seems plausible to expect *P. pinaster* to present more difficulties in post-fire reestablishment than *P. halepensis* and for the geographic range of *P. pinaster* to be reduced (Benito-Garzón et al. 2008).

6.2.5 Herbivory

In general, grazing activity in post-fire serotinous pine forests has been not different to that of other pine forests in the Mediterranean Basin. The grazing load depends on the cattle species and the success of the natural regeneration. Although in the first 2 years a great cover of legumes and other pasture species cover the burnt areas, in the case of serotinous pine forest, the advisable is to forbid it since 2 or 3 years after the fire to check how the forest regeneration is developing. If the natural regeneration is ensured, the maintenance of agricultural and pastoral activities can be used as tools for fire prevention.

The grazing rate depends on the plant community, the ecosystem response to fire (related to fire severity) and the livestock. Depending on the herbivore or lignivore and the livestock size (trampling effect), the shrub vegetation could be reduced and disappear mainly in areas bearing heavy and wild lignivores (McEvoy et al. 2006). Pine seedlings are compatible with horses and cattle but not with goats and sheeps which should be forbidden in regenerating areas after the fire (Mosquera et al. 2006). In other hand, because the source of more than 30% of the wildfires in southern Europe is agricultural and grazing activities, it is essential to consider the needs of rural communities who use fire for their livelihood, and how burning might help restore and maintain these ecosystems.

6.2.6 Pest and Diseases

Fire and insects are natural disturbance agents in many forest ecosystems, affecting succession, nutrient cycling and forest species composition. Abundance and species richness of herbivorous insects (xylophages and sap-feeders) decrease during succession whether or not major changes are found between the sites in terms of abundance or species richness of predators (Kaynaş and Gurkan 2008). Fire suppression policies implemented in the past (early 1900s) resulted in profound changes in biomass accumulation in forests, structure and species composition. Additionally, wildfires reduce some hosts that assist in the spread of pests and pathogens (McRae et al. 2001). Associated with these changes, there was an increasing vulnerability of forest stands to wildfires and damages during outbreaks of defoliating insects that are more frequent due to the increasing drought periods in the Mediterranean Basin. There is a gap in the knowledge of the relationship between serotinous pines and the outbreaks of the pest attacks; however, these pines are host of a large number of insects that can become pests (Mendel et al. 1985). The most common pest species in Mediterranean semiarid areas are bark beetles (*Carphoborus minimus, Hylurgus ligniperda, H. micklitzi, Hylastes linearis, Pityogenes calcaratus, Orthotomicus erosus, Tomicus destruens*), caterpillars (*Thaumetopoea pityocampa, Thaumetopoea wilkinsoni*) and other insects (*Neodiprion sertifer, Matsucoccus josephi, Palaeococcus fuscipennis*). Some studies that reforestation actions sometimes increase bark beetles attacks, mainly in semiarid areas. Also, some post-fire management measures, such as very early thinning, increase the sensitivity of these stands to pest attacks, such as *Pachyrhinus squamosus* in Spain (Gonzalez-Ochoa and de las Heras 2002). Outbreaks of *O. erosus* after the fire has been reported in *P. halepensis* stands in Israel (Mendel and Halperin 1982) and *Pinus pinaster* in South Africa (Baylis et al. 1986).

In *P. halepensis* and *P. brutia* stands the higher risk to become a pest was showed by Hymenoptera and Coleoptera families (Scolytidae, Buprestidae and Cerambycidae) in Greece (Markalas 1991). Pine engraver (*Ips sexdentatus*) and other scolytids (*Tomicus* spp, *Hylurgus piniperda*, *Hylastes* spp, *Pissodes notatus*) were the most frequent insects attacking fire-injured Maritime Pine trees in all burned areas in Spain (Vega et al. 2010).

6.2.7 Post-Fire Forest Conversion

In burned pine forests where reforestation is necessary or at least advised, a window of opportunity opens to introduce new forest structure and even, new species.

As previously mentioned one common conversion is a change towards a mixed pine-oak forest, with gains in biodiversity and ecosystem resilience (Valdecantos et al. 2009). Conversion can also include changing land cover to create more fire resistant and resilient landscapes, increasing heterogeneity and landscape barriers or filters that inhibit the spread of fire (see Chap. 1).

To convert an area into serotinous pine stand, the selection of the main tree species is advisable, taking into account that the optimal seed provenance selection is mandatory to ensure genetic adaptation and promote resilience.

6.2.8 Climate Change

The three serotinous pine tree species present in Mediterranean forests could fail to develop adaptive strategies to increased aridity or habitat fragmentation as has been recorded in the last two decades (Alig et al. 2002). Global change is also considered to increase the severity and recurrence of fires, the frequency of extreme events like hurricanes or windstorms, pest attacks and to promote invasions by alien species (Dale et al. 2001). Regarding European pines, and specifically the serotinous pines distributed in the Mediterranean Basin, there are some points in which climate change and its effects are impacting populations:

- Pine species which are not well adapted to intense drought or irregular weather, mainly during spring, are in jeopardy due to habitat reduction. In this way, *P. pinaster* distribution is becoming reduced, while altitudinal requirements are increasing and they are being replaced by other coniferous or broadleaf species (Resco et al. 2007).
- *P. halepensis* distribution is shifting upwards (up to 200 m) in the mountains close to the coast, replacing the lower altitudinal range of other species such as *P. sylvestris* in Southern France where the latter had lower productivity (Vennetier et al. 2005).

- Higher fire intensity and recurrence induced by climate change could reduce the period between two fire events, hampering the natural regeneration of the stand, altering the process of autosuccession (Initial Floristic Model) even in fire-prone ecosystems (Díaz-Delgado et al. 2002).
- Fire season is getting longer, which could decrease the resilience of communities and ecosystems adapted to recover after fire events in the drought season (Eugenio and Lloret 2004).
- The temperature and the longer activity period affect the survival of insects and the synchronization mechanism between hosts and herbivores. It alters diapauses, resulting in faster development and a higher feeding rate, increasing the risk of pest attack, mainly in the lower altitudinal areas, such as in the pine processionary moth (Battisti 2004).
- Higher aridity promotes an increase in respiration rate and decrease in Gross Primary Productivity. It promotes carbohydrate use for growth and survival in drought years, reducing the investment in reproduction which increases the immaturity risk in young Aleppo and East Mediterranean (or Anatolian) pine stands, which in turn depend on the stored canopy seed bank for regeneration (Keeley et al. 1999).

The impacts from global climate change and its associated disturbances have created a growing demand for scientific research.

6.3 Case Studies

6.3.1 Early Post-Fire Management in Eastern Mediterranean Basin

6.3.1.1 *Pinus brutia* Ten

A wildfire burned 1,664 ha in summer 1997 close to Thessaloniki, the North-western coast of the Aegean Sea (longitudes 22º57′ to 23º04′E; latitudes 40º35′ to 40º39′N). It was a reforestation coming from the fifties and resulted in a mixed coniferous forest composed mainly of Mediterranean pines and cypresses, although the climax vegetation in the area was an oak forest. The altitudes ranged between 85 and 560 m asl and the soil was shallow, rocky and intensely eroded with low productivity. It was an urban forest (about 60 years old) composed mainly of *P. brutia*.

After the fire, there was natural regeneration of *P. brutia* with high survival rates (72–90%) covering about 86% of the soil surface. The public administration decided to recover the stand as an urban forest managed for multiple use frameworks, such as flood prevention, soil protection and recreational use. An afforestation to increase diversity was carried out using two coniferous and two deciduous tree species but they showed high mortality rates (Grigoriadis et al. 2009).

The *P. brutia* tree density was very high and a thinning treatment was carried out 10 years after the fire. It improved tree growth, differentiation of young stands and shortened the age for cone production, which is very important due to the immaturity risk. In addition, the thinned stands showed higher mean diameter, mean height, vitality and faster developmental tendency compared to the unthinned stands. Autosuccession was occurring with the natural regeneration of *P. brutia* coming from restored areas, with similar results to those found in natural forests (Tsitsoni et al. 2004).

6.3.1.2 *Pinus halepensis* Mill.

A high recurrence of fires has been recorded in Kassandra peninsula (longitudes 25º25′ to 25º35′E; latitudes 39º90′ to 40º10′N), close to Chalkidiki (Northern Greece). The northern Kassandra peninsula is almost flat (100–250 m amsl). The soil is marly, covered by red-clays. Before fire, the forests were mainly managed for resin production. A large forest fire occurred in summer 1984 and the first plan to manage the burned area was no action and the short term monitoring of some plots to record the level of natural regeneration. Natural regeneration was supported by controlling grazing (goats and sheep were restricted to rangelands) and removing snags from late autumn to early winter in the year after the fire. On steep slopes (>50%), log dams were built with the burned branches to prevent soil erosion. In addition, new firebreak strips and roads were opened by removing the understory (Tsitsoni 1997) because a new fire within less than 10–15 years could be catastrophic for natural regeneration of *P. halepensis* (immaturity risk; Keeley et al. 1999).

The natural regeneration of *P. halepensis* in the first 8 years after the fire varied from 0.6 to 14.3 seedlings m^{-2}. The height of the Aleppo pine seedlings in their first year ranged from 1 to 30 cm (Tsitsoni 1997). Ten years after the fire, the plant composition in the burned area was following the Initial Floristic Model (Egler 1954). Shrub biomass accumulation was rapid during the early years and continued until the age of 10, reaching 11.5 t ha^{-1}. At this age the biomass was characterized by the high contribution of *Cistus* species (24.9%; Ganatsas et al. 2004).

A new fire occurred in summer 2006 in the same area, burning about 7,700 ha of Aleppo pine forest (Fig. 6.2). The emergency actions were to carry out salvage logging, using the deadwood to build log dams (148 km) and branch barriers (447 km) along the contours. Furthermore, in the second year after the fire, flood prevention tasks were planned and implemented using wooden dams (447 dams). Three years after the fire, a reforestation was carried out in places with no natural regeneration or very low tree density due to the high fire recurrence.

In nearby Penteli, in central Greece, two fires (summers 1995 and 1998) burned an Aleppo pine stand. The low fire intensity allowed for recovery of the composition of the vegetation but not the density. However, the main tree species, Aleppo pine, showed a very low natural regeneration in the monitored plots 3 or 4 years after the fire. Afforestation was necessary and the seedlings were planted in patches (2 × 2 m)

Fig. 6.2 Burned area in Kassandra Peninsula, Northern Greece, after the big wildfire in 2006. Upper image show the area a few weeks after the fire and the lower image the natural regeneration 3 years after the fire (summer 2009). (Source: Tsitsoni Thekla, Aristotle University of Thessaloniki)

using seeds from the same provenance, applying soil treatments (Tsitsoni 1997). Three restoration methods were used: two planting methods (paper-pot and bare-root seedlings), three seeding types (patches, strips and pits in lines) and no restoration action (Zagas et al. 2004). The results showed that all accelerated the rate of regeneration, while the most appropriate method to improve the regeneration process was to plant paper-pot or bare-root seedlings in pits which protected the seedlings.

The number of seedlings planted was reduced to a minimum (300 seedlings ha^{-1}), since soil protection is secured from the dense, broad-leaved evergreen vegetation (Zagas 1994). This choice has an additional economic advantage; planting seedlings in a similar density to mature stands implies no further silvicultural treatments should be needed if the seedling survival was successful.

6.3.2 Early Post-Fire management in Western Mediterranean Basin

6.3.2.1 *Pinus halepensis* Mill.

We studied three large fires which burned ca. 65,000 ha in July 1994, in Spain. In the burned forests, *P. halepensis* was the main tree species. All three areas naturally regenerated after the fire with very high initial seedling density, from about 7000 to 30,000 pine trees ha^{-1}. In the three areas, *P. halepensis* was the dominant regenerated species after fire (>90% of ground cover) and the soil was mainly carbonate substratum with a pH of about 8.5 and low slope (<5%).

Areas are described starting from the north and moving south:

- *Bages* in Barcelona (42°6′N, 2°1′E), 21,500 ha burned. Subhumid.
- *Yeste* in Albacete (38º20′ N, 2º20′ W), 14,000 ha burned. Dry.
- *Moratalla* in Murcia (38°16′, N1º38′W), 30,000 ha burned. Semiarid.

Immediately after the fires, several plots were set to monitor natural regeneration. Ten years later early silvicultural managements consisted of thinning the young Aleppo pine trees. In December 2003, a mixture of free and low thinning methods where applied, removing trees to control stand spacing and favour desired trees using criteria linked to the health appearance of individuals from the lower crown classes to favour those in the upper crown classes. The final pine tree density [AU4] achieved was about 1,600 pine trees ha^{-1}. Following Moya et al. (2008) and Verkaik and Espelta (2006), in spring 2004 six experimental plots were randomly established per site, and a preliminary experiment was conducted to compare the size of the canopy seed bank stored (number, test of viability and germination of seeds were included). The pines in the monitored plots were tagged and the total height, basal diameter (30 cm above the soil to avoid irregularities) and crown coverage were recorded. In addition, the different cohorts of cones were identified and counted, according to their colour and position in the canopy (Daskalakou and Thanos 1996) and the serotiny level was calculated following Goubitz et al. (2004). The first reproductive year (age when a pine tree begins to bear cones) and the number of cone-bearing trees (reproductive trees) were also recorded to characterise the reproductive stages (juvenile and reproductive phase).

Growth and reproductive characteristics of trees from three sites were compared to observe the effects of the treatments on pine growth, improvement of cone yield and increase in canopy seed bank (Table 6.1). We found that thinning improved growth and the amount of reproductive trees (>10%) just 1 year after to apply the silvicultural treatments. Opened cones were not recorded, implying the serotiny level did not vary from 2004 to 2005. Across the decreasing site quality, a decreasing number of reproductive trees and an increasing serotiny level which was not influenced by thinning were found.

The first reproductive year was found in pines 4–7 years old in all the study plots. The cone yield was improved to the individual tree-level by thinning, in both mature and serotinous cones. However, when the values were recorded for unit of surface (ha), we checked that the final amount of cones born and the canopy seed bank stored in them was lower in thinned plots due to the lower pine tree density. The final canopy seed bank was 37,000 ± 5,000, 154,684 ± 39,448 and 16,710 ± 4,592 seeds ha^{-1} in Bages, Yeste and Moratalla, respectively.

Initial tree density has been shown to have a direct relationship with growth and reproductive characteristics, mainly abortion and cone production, in different site quality areas (Moya et al. 2007, 2008b). Therefore, optimal management should be developed and designed to be flexible for diverse objectives specific to the area. Silvicultural treatments improve health and reproductive characteristics in these stands, although a drastic drop in pine tree density could require time to respond to the treatment. To improve the health of the stand and resilience to fire we recommend high intensity thinning in young stands with high tree density. These simple and cheap silvicultural treatments shape juvenile stands that regenerate after fires, improving the fertile seed production and thus fostering species resilience. At the same time, the treatment can be used as a prevention tool to reduce fire risk.

6.3.2.2 *Pinus pinaster* Aiton.

Two populations of *P. pinaster*, Nocedo (Lugo province) and Cabo Home (Pontevedra province) in north-western Spain were burned in the summers of 2000 and 2001, respectively (Table 6.2). Within each stand, five square plots (30 × 30 m) were set in each area (border length between plots was at least 15 m). We found different fire severity level (estimated from crown damage):

– *High*: trees with combusted crowns (crown fire).
– *Medium*: trees scorched (surface fire)
– *Low*: not scorched crowns (surface fire)

The burned trees were harvested some months after the fires in Nocedo (September 2001) and Cabo Home (March 2002). The logging slash treatments applied were: *clearcutting + slash chopping* (tractor with a mechanical chopper) in Cabo Home and *clearcutting + slash windrowing* (manual) in Nocedo.

To evaluate seedling density and mortality, 16 subplots (2 × 2 m) were set in each plot as a grid. On the first sampling date, all emergent *P. pinaster* seedlings were

Table 6.1 Reproductive and epidometric characteristics in sites naturally regenerated after fire [silvicultural treatments were carried out 10 years after the fire (winter 2004) and the study period extended for 1 year long]

SITE	Treatment	Density	Dbasal	H	CC	RP	MC	SC	SL
Bages	Control	56404	2.4 ± 0.2	212 ± 9	0.4 ± 0.1	52 ± 2	0.22 ± 0.05	2.20 ± 0.31	59
	Thinning	1655	3.3 ± 0.2	212 ± 11	0.8 ± 0.1	63 ± 7	2.27 ± 0.81	5.52 ± 1.46	
Yeste	Control	24211	3.1 ± 0.1	140 ± 4	1.2 ± 0.1	39 ± 8	1.47 ± 0.33	0.16 ± 0.06	89
	Thinning	1600	4.4 ± 0.2	177 ± 4	1.5 ± 0.1	49 ± 9	5.34 ± 0.79	3.64 ± 1.52	
Moratalla	Control	62669	1.6 ± 0.1	103 ± 3	0.6 ± 0.1	13 ± 5	0.04 ± 0.03	0.13 ± 0.05	100
	Thinning	1688	1.9 ± 0.1	111 ± 2	0.8 ± 0.1	26 ± 8	0.14 ± 0.09	0.16 ± 0.12	

Density Aleppo pine tree density (pines ha^{-1}), *Dbasal* basal diameter recorded 30 cm above the soil (cm), *H* total height (cm), *CC* Crown coverage of Aleppo pine trees (cm), *RP* reproductive pines (%), *MC* mature cones (cones $tree^{-1}$), *SC* serotinous cones (cones $tree^{-1}$), *SL* serotiny level (%)

Table 6.2 Description of the two studied maritime pine stands (*Pinus pinaster*) burned by wildfires in 2000 and 2001

Site	Fire date	Fire severity	BS	Age	SL	Climate	AR	MAT	T_{max}	T_{min}	Soil[a]
Nocedo	Summer 2000	high-medium	1050	45	0	Mediterranean	850	12.7	26.4	2	Alumi-dystric Leptosols
Cabo Home	Summer 2001	medium-low	32	53	79	Oceanic	1565	14	24.1	8.2	Dystric Cambisols

BS = burned surface (ha), *AGE* age of the stand before the fire (years), *SL* serotiny level (%), *AR* average rainfall (mm yr^{-1}), *MAT* mean annual temperature (°C), T_{max} mean of absolute maximum temperatures (°C), T_{min} mean of absolute minimum temperatures (°C)
[a]From Macías and Calvo (2001)

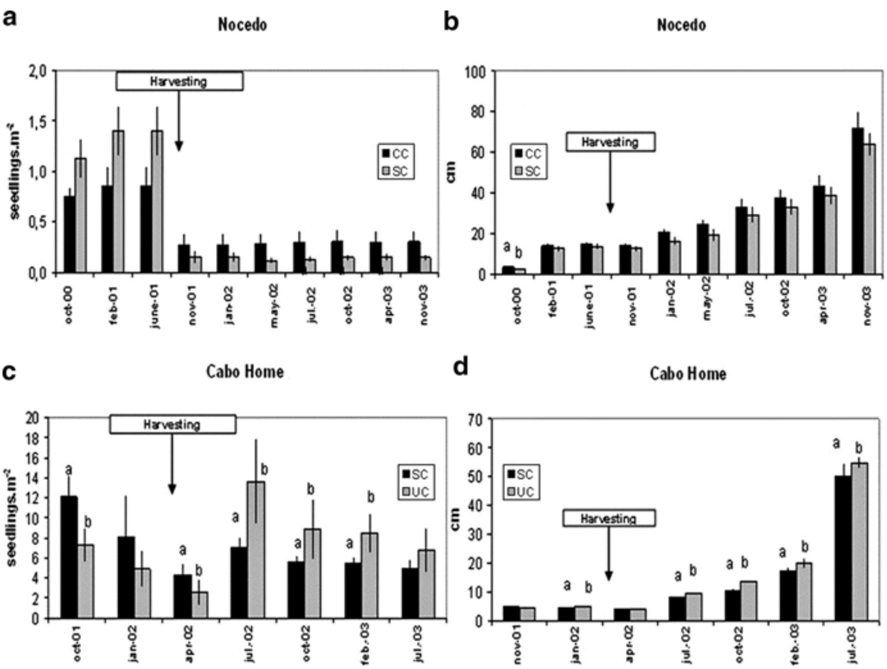

Fig. 6.3 Seedling density (number of seedlings) and seedling height (in cm) in two burned and treated maritime pine stands from the first autumn after the fire to 2003. *CC* combusted crowns, *SC* scorched crowns, *UC* unaffected crowns. Within evaluation dates means values indicated with different letter are significantly different ($P<0.05$). *Vertical bars* represent standard error. (Source: Cristina Fernandez, CIF Lourizan)

labelled and their heights were measured. In subsequent samplings, the number of dead seedlings was recorded and newly germinated seedlings were labelled. Seedling height was measured on each sampling date. In both stands, periodic measurements were carried out the first 2 years after the fire.

Seedling emergence began soon after the first rains in autumn in both sites. In both stands the initial seedling density was significantly higher in burned plots with lower fire severity. The salvage logging carried out 1 year after the fire induced a significant decrease in the initial seedling density. In Cabo Home, slash chopping favoured a new seedling cohort, especially in the unaffected crown plots, promoting an increase in seedling density the first spring after the fire (Fig. 6.3). Seedling height was higher in areas with lower fire severity in Nocedo, although not significantly. In Cabo Home, the mean seedling height was greater in the unburned plots (Fig. 6.3).

Scorched crown level exhibited higher seedling densities than trees with combusted crowns or unaffected crown in Nocedo and Cabo Home, respectively. This may be related to the fact that scorched trees dispersed seeds faster than trees with combusted crowns. In a study carried out in Cabo Home, Vega et al. (2010) found that seed release from scorched trees was faster than in non-scorched crowns. In burned *P. halepensis* and non-serotinous *P. pinaster* stands, the same recruitment pattern was observed (Martínez et al. 2002; Saracino et al. 1997).

The effects of harvesting on pine regeneration varied markedly depending on the stand and site characteristics. This suggests that it is not possible to design a common post-fire salvage logging schedule for all sites. For provenances with low serotiny, harvesting and logging must be planned and conducted more carefully. The combination of high temperatures and drought in the summer of 2001 in the south-facing slopes in Nocedo, and an early frost may have contributed to the sharp increase in mortality. By contrast, logging slash chopping in serotinous stands with unaffected crown trees can result in a new cohort of seedlings. This can be critical if the preceding one fails. Similar delayed pulse of germination was observed as a result of slash chopping in *P. pinaster* stands, by Canga et al. (2003) and Fernández et al. (2008).

There is no evident reason for higher seedlings in plots where fire severity, estimated through crown damage, was greater in Nocedo. The same conclusions were described for *P. halepensis* (Ne'eman 1997; Ne'eman et al. 2004; Pausas et al. 2003) and it was attributed to a greater nutrient availability and lower competition. Further research is necessary to clarify this point. Contrary to the results in Nocedo, in Cabo Home seedling height was greater in the unaffected crown plots than in scorched crown plots, but in this case, damping off as well as harvesting and slash chopping effects might explain those results.

From a management point of view, even in the harsh environment of the Nocedo stand, seedling stocking was sufficient to ensure an acceptable level of pine recruitment after wildfire although logging slash carried out in summer seems inadvisable.

6.3.2.3 Post-Fire Salvage Logging Impact on *P. pinaster* and *P. halepensis* Natural Regeneration

A literature searching of the studies on the impact of post-fire salvage logging on *P. pinaster* and *P. halepensis* natural regeneration was carried out. Nineteen researches with more than one replicate, available means and variances and including harvesting activities were included (Table 6.3). The method to calculate the size-effect was the logarithmic response ratio, using variance to weight the importance of each study (Kopper et al. 2009; Kalies et al. 2010). Random effects model was used in the analysis to check significant differences in mean response among categorical variables (type of slash manipulation). Between groups heterogeneity (Q) for each categorical variable followed a chi-square distribution which allows developing a significance test of the null hypothesis.

The level of serotiny explained a significant portion of the variability of the *P. pinaster* seedling density after fire (Gil et al. 2009; Vega et al. 2008a). Slash manipulation after clearcutting affected significantly to seedling density after harvesting, without effects of harvest intensities and slash disposal techniques. In some

Table 6.3 Summary of the studies used in the meta-analysis of impact of post-fire salvage logging on Mediterranean pine recruitment

Species	Reference	Site	Fire date	TT	SL	Treatment
Pinus halepensis	Saracino et al. (1993)	Italy (Taranto)	1988	0-24	–	EXTRACTION
	Martínez-Sánchez et al. (1999)	Spain (Albacete)	1994	10	–	CONTROL
						CLEARCUT+EXTRACT
Pinus pinaster	Madrigal et al. (2007)	Spain (Cáceres)	2003	2	High	CONTROL
						CLEARCUT+EXTRACT
						CLEARCUT+CHOPPING
	Fernández et al. (2008)	Spain (Ourense)	2003	13	20–33	CONTROL
						CLEARCUT+WINDROW
						CLEARCUT+CHOPPING
	Vega et al. (2008a)	Spain (Lugo)	2000	11	0	CLEARCUT+WINDROW (HIGH FIRE INTENSITY)
						CLEARCUT+WINDROW (LOW FIRE INTENSITY)
	Vega et al. (2008b)	Spain (Guadalajara)	2005	15	0	CONTROL
						CLEARCUT+WINDROW
	Gil et al. (2009)	Spain (Guadalajara)	2005	12	32.8	CLEARCUT+WINDROW (CONTROL SOIL)
						CLEARCUT+WINDROW (ALTERED SOIL)
					0.1	CLEARCUT+WINDROW (CONTROL SOIL)
						CLEARCUT+WINDROW (ALTERED SOIL)
	Madrigal et al. (2009)	Spain (Guadalajara)	2005	12	0	CLEARCUT+WINDROW
	Vega et al. (2010)	Spain (Guadalajara)	2001	6	75-83	CLEARCUT+CHOPPING (CROWN FIRE)
						CLEARCUT+CHOPPING (SURFACE FIRE)
	Castro et al. (2010)	Spain (Granada)	2005	8	-	CONTROL
						CLEARCUT+CHOPPING
						CLEARCUT

TT time to treatment (months), *SL* serotiny level (%), *WE* wood extraction, *CONTROL* no treatment was carried out cases, slash chopping favour a new seedling cohort, particularly in serotinous stands (Fernández et al. 2008; Vega et al. 2010), showing the least reduction of seedling stocking. Slash windrowing showed a significant negative effect, which could be influenced by the serotiny level although in some cases it is argued that this treatment could represent safe sites for pine installation (Castro et al. 2010).

Only a few studies found that salvage logging and slash manipulation could hamper pine regeneration, supporting that both ensured an adequate level of post-fire pine recruitment for the two species studied. It suggests that burned tree harvesting can be compatible with natural post-fire regeneration but low serotiny level and high water stress conditions can limit *P. pinaster* post-fire seedling establishment in many Mediterranean areas (Madrigal et al. 2010).

There is an obvious lack of information on the effect on salvage logging for serotinous pine species. Salvage logging might be detrimental or appropriated depending on serotiny level of the stand. For future studies we advise to include serotiny level, fire severity and a more appropriate control than the pre-treatment values in the sampling design.

Future studies should consider the origin of the stands because variability in provenance shows differences in serotiny level. Fire severity has also been shown to be a relevant variable in the regeneration process (Pausas et al. 2003; Broncano and Retana 2004; Vega et al. 2008a).

6.4 Key Messages

- Climate change influences community structure, species composition and the resilience to fire, even for serotinous pine forests adapted to fire.
- Serotiny has been documented as, although not exclusively, a fire adaptation and is defined as a morphological feature whereby the cones remain closed for years after seed maturation and open after exposition to high temperatures. However, it has been proved to be a more complex trait increasing the survival of released seeds which showed high biological quality (sound seeds, weight and germination rate) and higher heat insulation and resistance to fire.
- The serotinous pines in the Mediterranean area of southern Europe are the obligate seeders *Pinus halepensis*, *Pinus brutia* and *Pinus pinaster,* covering a wide range, mainly in xeric and lower altitudinal areas.
- These forests have usually been under-or over-managed depending on the economic value. Regarding post-fire management, salvage logging has been usual and after they have been given up until to reach the natural recovering. In the last decades, the management objectives are including ecosystem protection, water regulation, hazard prevention and the promotion of mature, diverse, and productive forests.
- Emergency actions are a main topic to protect soils and prevent erosion. The monitoring of burned areas to check the success of natural regeneration should indicate if no action, assisted natural regeneration or active restoration had to be carried out.

- Adaptive management should be developed to face up post-fire threats, such as soil loss, droughts, herbivore, pest attacks, etc., taking into account the variation induced by the predicted climate change.
- The case studies of post-fire management included in this chapter are focusing the forest management as a tool for restoration or assistance to natural regeneration. They show the effects of the main treatments used in burned serotinous pine stands in the Mediterranean area of southern Europe.

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