Chapter 5
Fire Ecology and Post-Fire Restoration Approaches in Southern European Forest Types

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5.1 Plant Adaptations to Fire and Post-Fire Response Mechanisms

Having suffered the repetitive action of fire in the course of their evolution, many plant species of Mediterranean environments have developed special adaptations to cope with it, ensuring their persistence in time. Plants possess two basic mechanisms to regenerate after fire: (i) vegetative regeneration (resprouting) of the same burned individuals, and (ii) establishment of new individuals through seed germination (Arianoutsou 1999; Bond and van Wilgen 1996; Whelan 1995). Knowing which mechanism exists in a given burned forest or shrubland is critical to evaluate the post-fire management alternatives.

Relatively few Mediterranean species do not show any specific regeneration mechanism after fire, and in this case the recovery of burned populations depends upon colonization from nearby unburned areas.

Resprouting after fire is a widespread trait in all fire prone environments and in all dicotyledonous plant lineages (Pausas and Keeley 2009). Resprouting occurs

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Fig. 5.1 Geophytes resprouting after fire from underground bulbs; *lower image*: *Urginea maritima*; *upper inner image Sternbergia* sp.; *upper image*: *Crocus* sp. (Source: Margarita Arianoutsou, Univ. of Athens)

usually at the root crown of the burned plants from dormant buds that remained intact after fire, being protected by the insulating soil. Resprouting is also the regeneration mechanism of plants that possess lignotubers, such as *Euphorbia acanthothamnos*, *Erica arborea*, *E. australis*, *E. multiflora*, or underground bulbs as many geophytes do, e.g. *Cyclamen* spp., *Muscari commosum*, *Urginea maritima*, *Crocus* spp. (Fig. 5.1).

Post-fire recovery of resprouters is a rather straightforward process as new shoot growth is supported by the almost intact belowground biomass surviving the fire.

Fig. 5.2 Woody species resprouting after fire from the root crown (*upper left image*). *Upper image on the right*: *Phlomis fruticosa*; *Lower image on the left*: *Quercus fraineto*. Often oaks can regenerate through epicormic growth, that is by developing new branches directly on the burned tree trunk (*lower image on the right*). (Source: Margarita Arianoutsou, Univ. of Athens)

Resprouts usually reach their reproductive maturity rather quickly, producing flowers and fruits a couple or, at most, a few years after resprouting. All oak trees (*Quercus* spp.) and most shrub species of phrygana and maquis all over the Mediterranean are resprouters (Fig. 5.2).

The species that have resprouting as their only regeneration mode after fire are called *obligatory resprouters*. *Facultative resprouters* are species that primarily regenerate through seed germination but they can also regenerate through resprouting, e.g. *Sarcopoterium spinosum* and *Erica* spp. Maquis species regenerate almost immediately after fire, while seasonal dimorphic species (phrygana) may resprout after the first autumn rains. This difference has been attributed to the different penetration depths of their root systems (Arianoutsou 1999). Resprouting is ensured as long as there is adequate storage of carbohydrates in the root crown, the lignotuber or the bulb (Jones and Laude 1960).

The second adaptation strategy for plant species to cope with fire is seedling recruitment. The pines of the thermo-Mediterranean zone (e.g. *Pinus halepensis* and *Pinus brutia*), most of the rockroses (Cistaceae) and many herbaceous leguminous species are *obligatory seeders* (Fig. 5.3). Seedlings emerge after the first autumn rains from seeds that were either dispersed before fire, having remained dormant in the soil as a soil seed bank, or dispersed after fire from a canopy seed bank.

Fig. 5.3 Obligate seeders. *Upper image*: (*left*) serotinous cone of *Pinus halepensis*; (*right*) *Pinus halepensis* seedling. *Lower image*: *Cistus creticus* seedlings massively appearing on the burned ground. (Source: Margarita Arianoutsou, Univ. of Athens)

Hard coated seeds normally lie dormant in the litter or topsoil layers and are released from dormancy by the heat shock induced by fire (Doussi and Thanos 1994; Ferrandis et al. 2001; Keeley and Fotherigham 2000; Papavassiliou and Arianoutsou 1993). Seeds forming these seed banks may also be stimulated by other factors related to fire cues, like high concentration of nitrates (Pérez-Fernández and Rodríguez-Echeverría 2003; Thanos and Rundel 1995), which is common in post-fire soils (Arianoutsou-Faraggitaki and Margaris 1982), change in the red/far-red ratio induced by canopy removal (Roy and Arianoutsou-Faraggitaki 1985) or smoke (Dixon et al. 1995; Pérez-Fernández and Rodríguez-Echeverría 2003).

Several pine species of the Mediterranean environments store their seeds in closed cones forming canopy seed banks; these pines are called *serotinous* (Fig. 5.3). High temperatures developing during fire on the plant canopy induce the dehiscence of the hard cones, the melting down of the resin keeping the scales of the cones tight and the subsequent seed dispersal (see Leone et al. 1999; Thanos and Daskalakou 2000; Ne'eman et al. 2004). Seed germination requires imbibition of the embryo and this takes place after the first autumn rains. Seedlings often appear in large numbers; however, high mortality usually occurs after the first dry season, which is mainly due to the drought effect (Arianoutsou and Margaris 1981; Papavassiliou and Arianoutsou 1993; Daskalakou and Thanos 2004). *Pinus halepensis* (Aleppo pine) and *Pinus brutia* (Brutia pine) are the most typical examples of Mediterranean serotinous pines.

Several plant species do not posses any specific post-fire adaptation mechanism, so once a fire occurs they can locally disappear. Typical examples are *Coridothymus capitatus* and *Juniperus phoenicea*. Similarly, non-serotinous pines, like *Pinus sylvestris* and *Pinus nigra,* once they are burned in an intense fire they are dependent for their recovery on seed dispersal from adjacent unburned patches. However, in the case of a light or moderate surface fire in which trees are protected by their bark and their canopy is not affected, seed germination may occur (Retana et al. 2002; Ordóñez et al. 2004; Arianoutsou et al. 2008, 2010a).

Ecological effects are shaped by fire regimes, namely the collective effects of fire frequency, intensity/severity, season, and size (Gill et al. 2002; Gill and Bradstock 2003). Fire frequency and severity are the most critical factors directly affecting plant responses. For the long term survival of the plants it is essential to know not only their adaptive traits towards a 'normally' occurring fire event, but also how they are affected by the fire regime. For example, there is an interplay between the capacity of species to survive and regenerate from fire and the interval between fires (fire recurrence). All species require a characteristic time length to replenish their regeneration capacity. Plants that are killed by fire and regenerate through seeds rely on this seed germination in order to persist at the specific location. For these plants, there must be sufficient time between successive fires for the seedlings to mature and produce seeds and hence replenishing soil and canopy seed banks. This time will vary between species that flower within the first post-fire year (such as herbaceous legumes), to those that may take 6–8 or more years to reach maturity (as pines do). If another fire occurs before these plants have matured, dramatic changes in the vegetation composition and physiognomy may occur (Arianoutsou et al. 2002, 2011). This is what is called the *immaturity risk*. The same holds for resprouters, when time between two consecutive fires is not adequate for them to replenish their carbohydrates reserves necessary for their post-fire regeneration. However, some species show extremely high capacity to resprout after frequent fires, beyond actual fire recurrence, as is the case of *Quercus coccifera* (Trabaud 1991a; Delitti et al. 2005).

Fire severity is a function of the amount of heat released by fire and the duration of the heating. It may determine the proportion of individuals that survive a particular fire, and it may also affect regeneration processes such as seed germination or resprouting potential. Fire-severity dependent mortality in resprouting species has been extensively documented for several species in the Iberian Peninsula (López-Soria and Castell 1992; Lloret and López-Soria 1993). Severe fires usually kill the stems of resprouters, but it seems that their regeneration at the population level is not generally affected. In relation to seeders, seeds lying in the soil seed bank seem not to be negatively influenced by intense fires. On the contrary, there are several references in the literature about heat induced seed germination after fire (e.g. Arianoutsou and Margaris 1981; Thanos and Georgiou 1988; Doussi and Thanos 1994; Keeley and Bond 1997).

5.2 Implications of Altered Fire Regimes Induced by Climate Change

Climate has a clear influence on fire regime (Pausas 2004). Consequently, if the expected changes in climate become a reality over the next century (IPCC 2007), an altered fire regime could have serious impacts upon Mediterranean ecosystems and their resilience towards fire (see also Chap. 11). Liu et al. (2010) have investigated the trend in global wildfire potential under the climate change due to the greenhouse effect. They measured fire potential by the Keetch-Byram Drought Index (KBDI), which is calculated using the observed maximum temperature and precipitation, and projected changes at the end of this century (2070–2100) by general circulation models (GCMs) for present and future climate conditions, respectively. Their analysis showed that future wildfire potential increases significantly in the United States, South America, central Asia, southern Europe, southern Africa, and Australia. Fire potential seems to move up by one level in these regions, from currently low to future moderate potential or from moderate to high potential. The largest relative changes were predicted for southern Europe. These findings are calling for increased and pro-active management efforts for preventing the potential catastrophic consequences and for ensuring an effective ecosystem recovery.

If fires recur more or less regularly, in the case of a climate change induced fire regime, selection pressure will favor those organisms that take advantage of the recurrence at a given interval and eliminate those that cannot follow (Flannigan et al. 2000). The vital attributes scheme developed by Noble and Slatyer (1980) has served as the basis of several predictive models to distinguish key groups of plant species that may be sensitive to changes in fire regime (Arianoutsou 1999, 2004; Arianoutsou et al. 2011; Kazanis and Arianoutsou 2004; Lloret and Vilà 2003; Pausas 1999; Pausas et al. 2004b). A key outcome of the vital attributes system is that differing functional types of plants will have differential sensitivity to recurrent disturbances such as fire. Functional types most sensitive to disturbance are those in which established individuals (adults and juveniles) are prone to death by fire (i.e. no capacity for vegetative recovery) and where seed banks may be depleted by fire. Consequently, sensitive functional types of this kind will be characterized by species that exhibit a high probability of mortality of juveniles and adults irrespective of fire intensity, plus seed bank types where germination is negatively affected by fire.

5.3 Fire Prone European Forest Types

In an overview of natural disturbances in European forests during the nineteenth and twentieth centuries, Schelhaas et al. (2003) estimated that forest fires were responsible for 16% of the total wood volume lost per year (35 million m^3), with larger damages caused only by storms (53%). The same authors have shown that during the period 1960–2000, most of the forest fires occurred in west Mediterranean, namely Spain and Portugal (44.9% of the total area burned), followed by the central Mediterranean (26.1%), mainly Italy and Slovenia, and east Mediterranean (17%), mainly Greece and Turkey. The remaining areas affected by wildfires included the sub-Atlantic (France mainly), with 7.3%, and the Central Pannonic (2.2%), mainly Poland and Romania.

These results highlight the significance of southern Europe as the major geographic area affected by wildfires in Europe. In terms of countries, Portugal, Spain, France, Italy, Greece and Turkey are the most affected ones (Schmuck et al. 2010), and forests in these countries are particularly fire prone. However, some countries in central and eastern Europe, in some years with dry and hot weather, have thousands of hectares burned (e.g. Bulgaria and Poland) (Schmuck et al. 2010).

Following the EEA (2007) forest type classification, there are two major forest categories – (a) broadleaved evergreen forests, and (b) coniferous forests of the Mediterranean, Anatolian and Macaronesian regions – occurring in the regions of Europe more affected by wildfires.

Broadleaved evergreen forests occur mainly in the thermo and meso-Mediterranean vegetation belt, whose climate determines the dominance of broadleaved sclerophyllous trees. The present distribution and physiognomy of Mediterranean evergreen oakwoods is the result of a long history of anthropogenic disturbance by clearance, coppicing, fires and overgrazing, resulting in vast areas covered today by degraded stages of evergreen oakwoods: arborescent matorral, maquis and garrigues (EEA 2007). The more fire prone types are the Mediterranean evergreen oak forests with *Quercus suber*, *Q. ilex, Q. rotundifolia* and *Q. coccifera* as the main species. The cork (*Q. suber*) and holm (*Q. ilex*/ *Q. rotundifolia*) oak forests, corresponding to the drier types, are the most widespread evergreen oak forests in the Mediterranean region and the more fire prone. Kermes oak (*Q. coccifera*) garrigues are another fire prone type. Among these *Quercus* species, the cork oak has the unique feature of having a bark with commercial interest (cork) that is exploited, which makes it very peculiar within the post-fire management context. This species is addressed in a separate chapter in this book (Chap. 9).

The coniferous forests of the Mediterranean, Anatolian, and Macaronesian regions is a broad category of conifer (pines, firs, junipers, cypress, cedar), mainly xerophytic, forest communities distributed throughout Europe (EEA 2007). The more fire prone coniferous forest types are the thermophilous pine forests with *Pinus pinea*, *P. pinaster*, *P. halepensis* and *P. brutia*, largely widespread in the lowlands of the circumediterranean region. These correspond to the forest types more affected by wildfires in Europe, in particular the three latter types. Mediterranean and Anatolian black pine (*Pinus nigra*) forests (and plantations), as well as Mediterranean and Anatolian Scots pine (*Pinus sylvestris*) forests in the mountain ranges of the Iberian peninsula and northern Greece, are also fire prone types in Southern Europe. However, in the post-fire management context, these pine forests differ from the former by the fact that they do not have serotinous pines (as *P. pinaster*, *P. halepensis* and *P. brutia* do) that enable the natural post-fire regeneration of these forests. These two broad types of pines (serotinous and non-serotinuous) are addressed in separate chapters (Chaps. 7 and 8).

Thermophilous deciduous forests are a third forest category that is also fire prone, although to a lesser extent. Thermophilous deciduous forests occur mainly in the supra-Mediterranean vegetation belt. Anthropogenic exploitation has modified the natural composition of these forests, leading in most cases to the elimination of species without commercial interest or with poor resprouting capacity or, conversely, the introduction of other forest species that would not occur naturally (chestnut) (EEA 2007). Within this category, the more fire prone types are probably the *Quercus pyrenaica*, *Quercus faginea*, and chestnut *Castanea sativa* forests

In addition to these native forests, plantations have increased in the last decades, mainly in some countries such as Portugal and Spain. Although these are considered a different forest type by EEA (2007), in this book they are pooled in the corresponding forest type, except in cases where the species used is exotic (e.g. eucalyptus plantations in Portugal). Independently of being cultivated or having invasive character, exotic species are an increasingly important post-fire management issue in some regions in Europe. This topic is examined in more detail in Chap. 10.

The total area burned per year is not exclusively composed of forests. Some non-forest land covers, and in particular shrublands, are highly susceptible to wildfires and represent a significant proportion of the total area burned (e.g. Pausas et al. 2008). Furthermore, increased fire frequency is turning former forests into shrublands, and promoting homogeneous landscapes covered by different shrubland types (Moreira et al. in press). The post-fire management of shrublands is dealt with in Chap. 12.

5.4 Major Questions in Mediterranean Forest Restoration

This section introduces some key issues in post-fire management which are common to all forest types. These include measures to promote soil protection, the management of burned trees, restoration or conversion, the use of active or indirect restoration, the management of herbivory, alien species, and pests and diseases.

5.4.1 Soil Protection to Reduce Erosion Risk

Soil erosion is among the most damaging post-fire processes. Soil degradation and erosion risk may be greatly enhanced by fires through the combined effect of direct soil heating and temporal loss of protective soil cover (Vallejo 1999). Water erosion may produce onsite loss of soil productivity and offsite siltation and flooding causing damages to humans and structures. Soil losses may be irreversible at the ecological time scales if they exceed soil formation rates, which are low in Mediterranean regions as in dry regions in general (Wakatsuki and Rasyidin 1992). Therefore, for ecological and safety reasons, reducing soil erosion and runoff risk should be the first priority in post-fire management (Vallejo and Alloza 1998; Vallejo 1999).

The major factors affecting soil erosion risk (Wischmeier and Smith 1978; Scott et al. 2009) are related to topography (slope grade and length), rainfall intensity, soil erodibility (related to soil properties), plant cover (including litter), and artificial erosion control measures like slope terracing. Fire may significantly affect soil erodibility, depending on fire severity, and, especially, plant cover, although both factors are partially interrelated: for low severity fires, plant cover and litter may partly remain thus protecting the soil from wind and water erosion. At the landscape and even at the hillslope scale (Schoennagel et al. 2008; Gimeno-García et al. 2011), fire severity is usually quite heterogeneous in space and not very high at the soil level for surface and crown fires which are the most common in the Mediterranean Basin. In severely burned areas, plant cover recovery rate is controlling how long after fire the soil will be exposed to the direct impact of raindrops, especially for heavy rains, and to excessive water erosion risk. Modelling these factors will allow identifying areas exposed to high soil erosion risk as a basis for planning post-fire soil protection actions (see Chap. 1 and Alloza and Vallejo 2006).

From the post-fire management perspective, the most critical factor affecting soil erosion risk which is susceptible of manipulation is plant/mulch cover. In Southern Europe, high intensity rainfall is most likely to occur by the end of summer and autumn. Therefore, for summer fires, the most common in the region, there is a high erosion risk right after the fire, when low plant cover and heavy rains may coincide in time (Vallejo 1999). The risk will continuously decrease as plant cover regenerates. Hence soil protection measures should be taken as soon as possible after the fire in areas where high erosion and runoff risk have been identified, and they have to be effective in the very short term. These measures are grouped under the concept of *post-fire emergency rehabilitation* (see Robichaud et al. 2000 for a critical review) or *emergency interventions* (see Chap. 1). The Forest Service of the United States Department of Agriculture has published a comprehensive catalogue of treatments for emergency response after forest fires (BAER, Napper 2006). Following the BAER procedures, the main topics to take into account when defining where to apply those emergency measures include 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 situation of infrastructures. An emergency plan should be set defining the priorities, time frame for implementation, personnel and funding availability, coordination of active agents (authorities, stakeholders and politicians), economical, social and environmental costs.

Land (hillslope) treatments stabilize the burned areas by preventing or mitigating the negative effects of fire. The most used techniques include mulching, erosion barriers (Fig. 5.4), scarification, slash spreading, planting and seeding, control of

Fig. 5.4 Forms of barriers to control post-fire soil erosion; *upper image*: branches barriers placed across a stream (Arkadia 2007 fire, Greece. Source Margarita Arianoutsou, Univ. of Athens); *middle image*: log barriers (Mt. Parnitha 2007 fire, Greece. Source Margarita Arianoutsou, Univ. of Athens); *lower image*: fine branches barriers, (Useres 2007 fire, Valencia Region, Spain. Source: Teresa Gimeno, CEAM)

invasive species and protection of special sites and habitats. Stream and channel treatments are used to reduce or mitigate water control and quality, trap sediment and debris and maintain stream and channel characteristics.

The application of an organic layer of mulch, either alone or combined with seeding native grasses, is an effective rehabilitation option on steep slopes with low plant cover and high erosion risk, as it is aimed at reducing rain splash, surface flow, 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 may be also effective post-fire management practices for reducing physical soil degradation and erosion.

5.4.1.1 Identification of the Conditions that Might Require Emergency Actions: Do They Relate to Forest Type?

Emergency treatments should be implemented in burned areas showing high erosion and runoff risk, with slow natural plant recovery rate, and when there are high values at risk downslope. Among these factors, the only that may be specific to a given forest type is the post-fire recovery rate of vegetation, as it depends on plant regeneration strategies (see Sect. 5.1). Hardwood forests usually resprout after fire, therefore if stand density is high enough, plant cover regeneration would quickly protect the soil in front of erosive agents.

For forests dominated by obligate-seeders, e.g. pine forests, plant cover regeneration in the short-term (up to 1–2 years after fire) mostly depends on understory vegetation, and this may vary within a forest type due to differences in land use and forest management history, stand age and density, soil characteristics, etc. For example, in Aleppo pine forests we can find a shrubby understory dominated by resprouters on limestone, with very low post-fire erosion risk, or an understory dominated by obligate seeders on old agricultural fields, with high erosion risk. Therefore, for pine forests, no species specific approach on post-fire emergency actions can be generalized.

5.4.2 Salvage Logging

After forest fires, one of the first decisions to take is how to manage the affected timber. Harvesting of commercially valuable dead or damaged trees (salvage logging) is the most common practice, provided burned timber has enough economic value to pay for the logging operations and to yield some benefit to the forest owner. Timber value continuously decreases as time passes after fire because of the wood decay, thus the forest owner is interested in logging as soon as possible to maximize economic benefit. In practice, for large fires it is almost impossible to rapidly harvest

Table 5.1 Argued pros and cons of salvage logging

Potential benefits	Potential negative impacts
To obtain some economic benefit of charred logs[a]	
To avoid boring insect pests[b] (e.g. Scolitydae)	
To improve pine germination (if logging is immediate) and avoid damage to regenerated pines[c]	Logging has detrimental effects on seedling growth[d] Microsites around burned trees favour regeneration and pine seedling germination[e] Salvage logging reduces forest breeding birds and their seed dispersal activity which is critical for late successional species[f]; also reduces deadwood associated fauna[g]
Trees naturally falling down (usually 2–3 years after fire) expose tree crown to erosion[h]	Dragging charred logs may produce soil surface degradation and soil erosion – rill erosion[i]
Risk of accidents by falling trees in inhabited areas	
To reduce landscape visual impact	

[a](SAF 1996); [b]Amman and Ryan (1991); [c] Roy (1956); [d]Gayoso and Iroumé (1991); Beghin et al. (2010); [e]Ne'eman (1997); Ne'eman and Izhaki (1998); Bautista and Vallejo (2002); [f]Castro et al. (2009); [g]Cobb et al. (2010); [h]Poff (1989); [i]Terry (1994); Bautista et al. (2004)

all burned forest area, therefore priorities should be established and this gives time for planning and allows for introducing ecosystem conservation criteria, in addition to the immediate economic value.

Salvage logging is controversial (Lindenmayer et al. 2004). Table 5.1 provides a list of the most commonly argued pros and cons of conducting salvage logging as soon as possible after a fire.

Economic benefits of salvage logging depend on the timber quality, distance from roads, market conditions, and forest ownership. Megafires may saturate charred wood market. Usually, private owners have more interest in taking some revenue from charred wood, and in the European Union they often receive subsidies from forest administrations for salvage logging, on the basis of its assumed beneficial ecological impacts to promote forest recovery.

Some of the potential ecological benefits and/or impacts of salvage logging may largely depend on the specific forest site and fire characteristics (Peterson et al. 2009). However, there is a tendency among forest managers to overgeneralize the pros and cons of logging. For example, charred wood potential to trigger a pest outbreak may depend on the specific threatening insect species biology and on the degree of damage of the burned trees (see Sect. 5.4.7).

In relation to soil erosion, shortly after fire the ash layer and the unprotected soil are highly sensitive to trampling and mechanical operations. Thus, short-term salvage logging on vulnerable soils may cause more erosion than the fire itself.

In Valencia, eastern Spain, Bautista et al. (2004) recorded up to 50 Mg ha^{-1} $year^{-1}$ soil loss in rills during the first 3 years after logging on *Pinus pinaster* stands developed over sandstones. Post-fire erosion rates in the same region were less than one order of magnitude lower (Mayor et al. 2007; Llovet et al. 2009). However, in the same study, Bautista et al. (2004) found that logging on stands developed over limestone produced low erosion rates.

Obviously, the impact of logging on soil erosion depends on soil erodibility and topography. For vulnerable soils, logging techniques avoiding log dragging could be applied, such as using a cable yarder or an helicopter, though these techniques are expensive and difficult to justify from an economic point of view in the commonly low stocking Mediterranean forests.

In the region of Valencia, after some 20 years of post-fire monitoring, the following generic recommendations were proposed in unpublished technical reports (CMA 1994), further elaborated by Bautista el al. (2004):

1. Avoid short-term salvage logging on vulnerable soils, at least until a protective vegetation cover has developed (usually after the first post-fire spring). Therefore, logging should be planned according to ecosystems vulnerability, in the same line as any other post-fire management activity (see Chap. 1).
2. In any case, retain some snags in order to keep forest nesting birds and other biodiversity components.
3. Selective logging in patches could combine economic and ecological benefits, avoiding sensitive soil erosion areas and generating mosaics that would optimize biodiversity (Izhaki and Adar 1997).
4. Monitor surviving weakened trees for the risk of pest outbreaks.
5. Use branches as barriers or to produce mulching material (chipped wood) to protect spots with high erosion risk (e.g. gullies, road talus, see Fig. 5.4).

5.4.3 Forest Restoration or Forest Conversion

Stand replacing fires offer the opportunity to consider alternative forest types to be promoted (Moreira and Vallejo 2009) in the context of land planning. Stand replacing fires in the Mediterranean may occur mostly in the case of conifer forests, when fire affects young, immature stands, or species not having specific post-fire regeneration mechanisms (see Sect. 5.1), or when long dry conditions occur during the first vegetative seasons after fire. For hardwoods, changing the dominant tree species is difficult even after a fire, as it may require uprooting the stumps. This has been practiced sometimes to eliminate eucalypt forests and recuperate native oaks in the Iberian Peninsula, but with high economic investment and ecological impact on the soil.

Forest conversion could be considered for breaking the horizontal continuity in homogeneous forests, usually planted pine forests. These are very abundant in the Mediterranean countries owing to the large plantations conducted along the twentieth century (Vallejo 2005). Changing burned pine forest patches into hardwood forest would in many cases re-naturalize the area, increase gamma diversity and fire resilience, and reduce pest outbreaks. In addition, for fire management planning, introducing patches of different forests types (e.g. promoting riparian vegetation along creeks) or even other land uses (e.g. agricultural areas or pastures) located in strategic areas may reduce fire propagation risk and help fire suppression (see also Chap. 1).

One other situation where conversion may be a suitable alternative is when the composition of fire prone shrublands is changed by planting fire-resistant resprouting trees (see Chap. 12).

5.4.4 Active Versus Indirect Restoration

One major decision in the post-fire management of burned forests, when the restoration of the former forest type is the main objective, is whether to use natural regeneration (indirect restoration), if it is present or predicted to occur, or active restoration (plantations and seeding).

There is strong political pressure for reforesting or afforesting burned areas in the Mediterranean region as soon as possible after a wildfire, and this has been a common practice since the late nineteenth century, particularly in conifer forests (Pausas et al. 2004a; Vallejo 2005). As an example, following the 2006 wildfires in Galicia (Spain), which burned 150,000 ha of land, reforestation has been considered a restoration priority (Amil 2007). In the case of Portugal, policies for the reforestation of burned forests have been common in the last decades (Carvalho Mendes 2006). However, reforestation/afforestation may not be the best alternative in many cases and the current political and social paradigm of “compensating” areas burned with active reforestations in a similar or higher number of hectares should be changed (see Chap. 1).

Reforestations are usually carried out through active restoration techniques such as plantation or direct seeding (Lamb and Gilmour 2003; Vallejo et al. 2006). Planting is expensive due to the costs associated with the acquisition of plant seedlings from nurseries, their transport to the burned area (particularly in areas with difficult access), site preparation (usually this is the highest cost), and other costs associated with equipment, fertilizers, tree shelters, replacement of dead seedlings, and human labour. Furthermore, activities associated with soil preparation for planting may increase the risk of soil erosion, and the mortality rates of planted seedlings, although quite variable, are often higher than 50% (e.g. Maestre and Cortina 2004; Pausas et al. 2004a; Vallejo 2005). Direct seeding is less costly and can be applied in extensive areas (e.g. aerial seeding), but the success (seed germination and plant establishment) is also usually very low (e.g. Vallejo and Alloza 1998; Espelta et al. 2003a; Pausas et al. 2004a). Therefore, these active techniques have a low cost-effectiveness and should be considered only when other options are not feasible, e.g. in areas where no natural tree regeneration is expected and where there are no mature trees in the vicinity that might provide seeds to naturally colonise the site in the medium term.

As an alternative to active restoration, other applications of financial and human resources may be much more effective, in particular through exploring the potential of natural regeneration characteristic of many Mediterranean species, i.e. taking advantage of regeneration from seeds left in the ground by the burned vegetation (e.g. Pausas et al. 2004b; Holz and Placci 2005), or from resprouting of burned trees and shrubs (e.g. Espelta et al. 2003b; Lloret et al. 2005). In addition to the lower costs of these passive (natural) restoration techniques (e.g. Lamb and Gilmour 2003; Whisenant 2005; Vallejo et al. 2006), plant survival and growth rates are higher when compared to active restoration and consequently a higher and faster-growing vegetation cover is achieved. Sprouts have many potential advantages over seedlings or planted trees because they have an already established root system and high stored energy reserves, which may confer greater chances of plant survival and recovery. For example, Moreira et al. (2009) have shown that oak *Quercus faginea* and ash *Fraxinus angustifolia* resprouts in burned areas in central Portugal could survive 20% more and grew 4–5 times faster when compared to planted seedlings (see Chap. 1). The use of plant resprouting ability is already acknowledged as a powerful tool to restore ecosystems such as tropical dry forests (Vieira and Scariot 2006) or the Atlantic rainforests of Brazil (Simões and Marques 2007) and this should be extended to Mediterranean ecosystems. Thus, we advocate a much more frequent use of assisted natural restoration, based on the management of natural regeneration from seeds or resprouts. Depending on the objectives, this may involve thinning, the selection of shoots, and the control of unwanted vegetation. The costs associated with assisted natural regeneration can be much lower when compared to active restoration, meaning that with a similar amount of funding available a much larger area can be effectively treated. Of course the decision of opting by active or natural restoration will be constrained by the type of pre-fire vegetation, the ecosystem response to fire, and the objectives for the burned area.

5.4.5 The Management of Herbivory

Grazing animals (both domestic and wild herbivores) can be either beneficial or detrimental, depending on the post-fire management objectives.

The beneficial effects of grazing and browsing are that they contribute to reduce fuel loads and, thus, fire hazard (e.g. Nader et al. 2007; Tsiouvaras et al. 1989). In fact, fire and grazing are quite similar disturbances in several aspects of their impact on vegetation (Bond and Keeley 2005). Using animals (mainly domestic animals) in fuel management (what is sometimes called *prescribed herbivory*) requires knowing their feeding strategy. Browsers consuming woody species (shrubs and tree branches), e.g. goats or deer, are better suited for controlling shrubby areas than grazers consuming mostly herbaceous vegetation (e.g. cows or sheep). There are a few studies in southern Europe confirming this positive impact of grazing. For example, in a study in Spain, Valderrábano and Torrano (2000) showed that goats were effective in reducing the survival and growth of an invasive shrub with high ignition capacity. So, grazing animals can be used in post-fire management when the objective is to reduce fuel loads (e.g. in fuel breaks, or in areas at the wildland-urban interface). In some countries (e.g. Portugal, USA), there are commercial enterprises selling prescribed herbivory, with animals being transported and confined to specific target areas. Important factors to take into account to increase the effectiveness of grazing include: (i) the selection of the animal species, (ii) the selection of the grazing season and grazing period, and (iii) the establishment of an appropriate stocking rate. Pastoral fire has been practiced in the Mediterranean for thousands of years to suppress the unpalatable woody species to animals in favor of the herbaceous plants palatable to sheep and cattle (Blondel and Aronson 1999). The coordinated use of this old management tool can also greatly contribute to reduce fire hazard (Rego et al. 2010).

Herbivory can, however, be an important limiting factor when the pos-fire objective is the regeneration of burned areas (Vallejo et al. 2006). After a fire, the regenerating plants are particularly susceptible to animal consumption. The first species to emerge typically have high digestibility and are very attractive to herbivores (Hobbs et al. 1991; Hobbs 2006). Resprouting species support well a slight consumption, but can be affected if there is repeated consumption of their terminal buds, essential to their growth in height. For example, Catry et al. (2007) showed that deer browsing can have a strong negative impact on post-fire basal sprout regeneration of several Mediterranean broadleaved species. Seedlings will obviously die if consumed by animals. Grazing damage is not restricted to browsing, but also trampling. So, if an area is being managed for natural regeneration, or has been planted, and if there is a density of grazers compromising the success of this regeneration, managers will probably have to invest in the protection of regenerating/planted plants. This can be done using three main different alternatives: (i) reducing the animal population size; (ii) protecting individual plants; (iii) protecting areas from herbivores. The reduction of animal densities to levels that are compatible with plant regeneration could be the ideal solution from the vegetation recovery point of view. However, this may not be feasible or compatible with the management objectives, e.g. in areas with wild herbivores where hunting is not allowed. The protection of individual trees is adopted in many countries, regardless of fire occurrence, when animals have access to regeneration areas or plantations. Various types of protections of variable prices and efficiency are available. The most common approach is to protect each tree with a protective cylindrical-shaped wire mesh. Another possible protection method involves the application of chemical repellent but in most cases its effectiveness is short-lived or is still unproved. Fencing parts of the area to regenerate may be also a good option, depending on the size of the area to protect and on tree density. Generally for larger areas and higher tree density, this technique is cheaper than the protection of individual plants. This option has the added advantage of allowing a denser regeneration in the burned area, and a better ground cover, contributing to preventing post-fire soil erosion. Possible disadvantages include the limited access to the area and higher fuel accumulation that may increase fire danger. Temporary protection by electric fencing is most suitable for domestic species.

Finally, the use of nurse plants has been shown to provide protection from grazers in a study in Spain (Castro et al. 2002), and this topic deserves further research.

5.4.6 Fire and Alien Species

Alien species may be an important post-fire management issue in situations where fire promotes their occurrence, particularly if they become invasive and compromise the regeneration of native vegetation. Besides their impact on native diversity, if these new plant communities have different fuel properties that increase their combustibility then a feedback loop may be created where fire promotes invasive species that turn the affected area more fire prone (Brooks et al. 2004).

Most Mediterranean environments have experienced a high degree of human interference and disturbance, a process that dates back over 10,000 years, and this has resulted in a marked transformation of the vegetation (Heywood 1995; Thompson 2005). In contrast to other Mediterranean regions of the World, e.g. California and South Africa, where large areas of relatively intact vegetation still remain, much of the Mediterranean Basin has been transformed from its native state (Mooney 1988). The result is that Mediterranean plant communities are rather resistant to biological invasions as native species are likely to be good competitors under the strong selection regime imposed by humans on the Mediterranean flora and that the multiple stresses of drought, fire and grazing present a limitation to prospective alien plant species to establish and further to become invasive (Hulme et al. 2007, Arianoutsou et al., in preparation). For example, very few alien annual species have been recorded in early post-fire communities of *Pinus halepensis* forests in Greece. These taxa represent less than 1% of the regenerating flora and their abundance was not substantial (Kazanis 2005). So, despite the increasing pace of research related to biological invasions in the Mediterranean region (e.g. Vilà et al. 2007; Celesti-Grapow et al. 2009, 2010; Arianoutsou et al. 2010b) very few, if any, paper relates biological invasions to fire, in contrast with other regions of the world, and especially California.

The abundance of an exotic species, *Cortaderia selloana* (Pampas grass), has been related to potential fire occurrence in the Mediterranean (Doménech et al. 2005). *C. selloana* is a South American longlived perennial grass native to Argentina, Brazil and Uruguay which is considered invasive worldwide and it was first introduced as ornamental to Europe between 1775 and 1862 (Bossard et al. 2000). It has escaped from cultivation and it is invading abandoned farmlands, roadsides, shrublands and wetlands. *C. selloana* is considered to increase fire hazard because of the accumulation of dry leaves and flowering stalks on the plant.

Although fire usually opens temporarily the plant canopy in many forest and shrubland ecosystems offering empty space for colonizing species (Arianoutsou et al. 2011), it seems that alien species, if they are available in the vicinity, cannot easily compete with natives to become established in the burned Mediterranean habitats as it seems to be the case for other Mediterranean climate regions, e.g. California (Keeley et al. 2005; Zouhar et al. 2008), Chile (Gómez-González et al. 2011), South Africa (van Wilgen and Richardson 1985; van Wilgen et al. 2010) or Australia (Thomson and Leishman 2005; Miller et al. 2010). However, there is some evidence that in areas of Southern Europe with less dry climates, invasive species are becoming a problem in burned areas. In northern and central Portugal, for example, the genera *Acacia*, *Hakea*, *Ailanthus* and *Eucalyptus* are a growing concern for forest managers, as their prevalence in burned areas is notoriously increasing (Catry et al. 2010).

It has been suggested that plant communities under altered fire regimes are more susceptible to invasion than those under a natural (historical) fire regime (Trabaud 1991b; D'Antonio 2000). For example, in most Mediterranean climate ecosystems, fire has been an important selective force shaping adaptive traits in native plant species (e.g. Pausas et al. 2006). Under conditions of natural fire frequency, the ability of native species to cope with fire leads to a relatively high ecosystem resilience to invasion, because alien species that colonize open areas are rapidly excluded from the system (Trabaud 1991b; Keeley 2001). But this dominance of native relative to alien species may be changed under a different fire regime and under different climatic conditions.

In conclusion, fire does not seem to promote the occurrence of exotic plants in most of the Mediterranean ecosystems, as these systems are rather resilient to disturbance and although aliens seem to prefer disturbed places to establish, they cannot cope with native species in such dry environments. It is true, that in most cases where alien plants have been recorded in natural systems, their presence was mostly concentrated in mesic places where nutrient availability is often higher (Stohlgren et al. 2003; Vilà et al. 2007; Arianoutsou et al. 2010c). This may explain why in some specific regions with less xeric environments, such as central and northern Portugal, exotic species are becoming a major problem in post-fire management (see Chap. 10). Clearly, more research is needed on this topic.

5.4.7 Pests and Diseases

Fire and insects are natural disturbance agents in many forest ecosystems, affecting succession, nutrient cycling and forest species composition. There are two basic mechanisms through which fire and pests or pathogens interact. One is the mortality or weakening of trees by fire and the subsequent promotion of damaging insect and pathogen populations. The second is through the mortality of trees caused by these agents, which contributes to dead fuel accumulation and increased fire hazard. However, there is not much information available on the relationships between wildfires and tree insect pests and pathogens in Southern Europe. Most research is focused on bark beetles and wood boring insects (e.g. Fernandez et al. 1999; Fernandez 2006).

Pines are the target of a variety of bark beetles that can cause tree death, branch dieback and reduced productivity (FAO 2009). Pine beetles of the subfamily Scolytinae (mainly the genus *Ips*, *Orthotomicus* and *Tomicus*) are considered a major problem in burned pine forests, but there is contradictory information on how to manage burned trees to minimize this hazard. For example, while forest managers often consider burned trees as the more likely to attract bark beetles, and partially affected trees (crowns partially scorched or consumed) are left hoping that they survive, in a region of Spain, Bautista et al. (2004) have shown that the latter are the ones that have the higher degree of infection thus should the first ones to log. In Northern Spain, Fernandéz (2006) also found that all *P. pinaster* trees with 25–50% burnt crowns were the first to be attacked by *Ips sexdentatus* and consequently die. The black trees (100% burnt crown) were mainly used for the establishment of secondary xylophagous beetles, such as Buprestidae and Cerambycidae. She concluded, however, that the most effective way to reduce the risk of mortality in healthy standing trees was to remove nearby dead and dying trees before the *I. sexdentatus* population grew large enough to attack healthy, less injured or recovering trees.

The moths of the genus *Lymantria* are significant defoliators of a wide range of broadleaf and conifer trees. Severe outbreaks can occur resulting in severe defoliation, growth loss, dieback and sometimes tree mortality. The pine processionary caterpillar *Thaumetopoea pityocampa* is considered the most destructive forest insect pest throughout the Mediterranean Basin. It is a tent-making caterpillar that feeds gregariously and defoliates various species of pine and cedar (FAO 2009). These species may cause intense defoliation with consequent tree weakness and even death. This might contribute to fuel accumulation, increasing fire hazard.

Some insect pests cause problems in specific geographical regions or forest types. For example, eucalyptus longhorned borers of the genus *Phoracanta* (Coleoptera: Cerambycidae) are serious borer pests of eucalypts, particularly those planted outside their natural range (FAO 2009), e.g. the eucalyptus plantations in Portugal. *Phoracantha* species tend to attack damaged or stressed trees; vigorous, well-watered trees are rarely attacked though it does occur. *Gonipterus scutellatus* (Coleoptera: Curculinoidae) is an exotic weevil infesting eucalypt plantations. The nematode *Bursaphelenchus xylophilus* (pine wilt nematode) was found in Portugal for the first time in 1999 and is a major threat for the maritime pine *Pinus pinaster* forests. The spread of this nematode is via wood-boring beetles of the genus *Monochamus*. They become infested with the nematode just before emerging as adults from diseased host trees. Adult beetles can act as vectors for thousands of nematodes (FAO 2009). In a recent review, Lindner et al. (2010) mention the likely increase in the virulence of thermophilic pathogen species in the Mediterranean, in response to climate change. Trees weakened by fire will become more susceptible, thus increased problems are expected. On the other hand, trees weakened or killed by these pathogens may increase fire hazard (Dios et al. 2007).

The fungi *Armillaria* spp. are a common worldwide pathogen of trees, woody shrubs and herbaceous plants causing root rot, root-collar rot and butt rot (FAO 2009). They cause wood decay, growth reduction and mortality, particularly in trees stressed by other factors. Climate change may be implicated in the increasing incidence of oak declines due to *Phytophtora* spp. *Phytophtora* require wet soil conditions to proliferate. In the last few decades floods have occurred more frequently creating favourable conditions for pathogen proliferation in forests. These floods have been followed by drought events that have weakened the trees and made them more susceptible to the pathogen, resulting in higher mortality than ever before (FAO 2009). As with other conifer bark beetle species, *Ips sexdentatus* is a vector for blue-stain fungi (*Ophiostoma* spp.) which hastens the death of trees, discolours the wood and can result in loss of timber value. Fire events by favouring the build-up of bark beetle populations may in particular activate this fungi-insect association (Bueno et al. 2010).

5.5 Key Messages

- Most Mediterranean plant species have developed regeneration strategies allowing their efficient recovery after fire. According to these strategies, species are grouped in obligatory resprouters, facultative resprouters, and obligatory seeders. Some species do not show any specific post-fire regeneration mechanism, hence they locally disappear after fire and can only re-colonize burned areas from adjacent unburned patches.
- Climate change projections indicate that wildfire potential will increase in Southern Europe, and this could reduce Mediterranean ecosystems resilience to fire.
- The most fire-affected vegetation types in Europe are thermophilous pine forests with *Pinus pinaster*, *P. halepensis* and *P. brutia*, broadleaved evergreen forests with *Quercus suber*, *Q. ilex*, *Q. rotundifolia* and *Q. coccifera*, and shrublands.
- Post-fire emergency rehabilitation actions should be applied to burned forests showing high erosion and runoff risk, with slow natural plant recovery rate. The forests more prone to fire erosion are pine forests with an understory dominated by obligate seeders. Mulching is one of the more effective techniques to decrease erosion risk.
- Salvage logging has economic and ecological benefits but also negative impacts, depending on the local conditions. General recommendations are: avoiding dragging logs in vulnerable soils, retain some snags for biodiversity purposes, monitor surviving weakened trees for the risk of pest outbreaks, and use charred wood for soil protection where there is high erosion risk.
- Forest conversion after fire could be considered for fire prevention and for the re-naturalization of the landscape.
- Although active reforestation/afforestation is the usual action taken by policy makers and forest administrations, in many cases assisted natural restoration is more efficient and cost-effective.
- Grazing animals contribute to reduce fuel load and fire hazard. However, herbivores may also limit the post-fire vegetation recovery. Therefore, protection actions should be taken where domestic and wild herbivore populations threaten regeneration.
- Alien invasive species are not much favored by fire in xeric Mediterranean environments, but in some moister areas they are becoming an increasing problem for post-fire management.
- Pests and diseases may increase dead fuel accumulation, and consequently fire hazard. Fire may facilitate pest outbreak, especially for pine bark beetles.

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References

Alloza JA, Vallejo VR (2006) Restoration of burned areas in forest management plans. In: Kepner WG, Rubio JL, Mouat DA, Pedrazzini F (eds) Desertification in the Mediterranean region: a security issue. Springer, Dordrecht

Amil ML (2007) Forest fires in Galicia (Spain): threats and challenges for the future. J Forest Econ 13:1–5

Amman GD, Ryan KC (1991) Insect infestation of fire-injured trees in the greater Yellowstone area. Res. Note INT-398, USDA, Forest Service, Intermountain Research Station, Ogden

Arianoutsou M (1999) Effects of fire on vegetation demography. International symposium on forest fires: needs and innovations, (DELFI), Athens

Arianoutsou M (2004) Predicting the post-fire regeneration and resilience of Mediterranean plant communities. In: Arianoutsou M, Papanastasis VP (eds) Ecology, conservation and management of Mediterranean climate ecosystems of the world. Proceedings of the MEDECOS 10th International Conference, Rhodes, Greece, Millpress, Rotterdam, Electronic Edition

Arianoutsou M, Margaris NS (1981) Producers and the fire cycle in a phryganic ecosystem. In: Margaris NS, Mooney HA (eds) Components of productivity in Mediterranean climate regions – basic and applied aspects. Dr W. Junk, The Hague

Arianoutsou M, Kazanis D, Kokkoris Y, Skourou P (2002) Land-use interactions with fire in Mediterranean *Pinus halepensis* landscapes of Greece: patterns of biodiversity. In: Viegas DX (ed) Proceedings of the IV International Forest Fire Research Conference, Millpress, electronic edition, (2002)

Arianoutsou M, Kazanis D, Copanellou I (2008) Report on the research and the study of the post-fire regeneration of the mixed pine forest of Strofylia, at the area of lake Kaiafa (GR 2330005, NATURA 2000). University of Athens, Department of Ecology and Systematics, (in Greek)

Arianoutsou M, Christopoulou A, Tountas Th, Ganou E, Kazanis D, Bazos I, Kokkoris I (2010a) Effects of fire on high altitude coniferous forests of Greece. In: Viegas DX (ed) Book of proceedings of the VIth international conference on forest fire research, Coimbra, Portugal (electronic edition)

Arianoutsou M, Bazos I, Delipetrou P, Kokkoris Y (2010b) The alien flora of Greece: taxonomy, life traits and habitat preferences. Biol Invasions 12:3525–3549. doi:10.1007/s10530-010-9749-0

Arianoutsou M, Delipetrou P, Celesti-Grapow L, Basnou C, Bazos I, Kokkoris Y, Blasi C, Vilà M (2010c) Comparing naturalized alien plants and recipient habitats across an east–west gradient in the Mediterranean Basin. J Biogeogr 37:1811–1823. doi:10.1111/j.1365-2699.2010.02324.x

Arianoutsou M, Koukoulas S, Kazanis D (2011) Evaluating post-fire forest resilience using GIS and multi-criteria analysis: an example from Cape Sounion National Park, Greece. Environ Manag 47:384–397. doi:10.1007/s00267-011-9614-7

Arianoutsou-Faraggitaki M, Margaris NS (1982) Decomposers and the fire cycle in a phryganic (East Mediterranean) ecosystem. Microb Ecol 8:91–98

Bautista S, Vallejo VR (2002) Spatial variation of post-fire plant recovery in Aleppo pine forest. In: Trabaud L, Prodon R (eds) Fire and biological processes. Backhuys Publishers, Leiden

Bautista S, Gimeno T, Mayor A, Gallego D (2004) El tratamiento de la madera quemada tras los incendios forestales. In: Vallejo VR, Alloza JA (eds) Avances en el estudio de la gestión del monte Mediterráneo. Fundación CEAM, Valencia

Beghin R, Lingua E, Garbarino M, Lonati M, Bovio G, Motta R, Marzano R (2010) *Pinus sylvestris* forest regeneration under different post-fire restoration practices in the northwestern Italian Alps. Ecol Eng 36:1365–1372

Blondel J, Aronson J (1999) Biology and wildlife of the Mediterranean region. Oxford University Press, Oxford

Bond WJ, Keeley JE (2005) Fire as a global "herbivore": the ecology and evolution of flammable ecosystems. Trends Ecol Evol 20:387–394

Bond WJ, van Wilgen BW (1996) Fire and plants. Chapman and Hall, London

Bossard CC, Randall JM, Hshousky MC (2000) Invasive plants of California's wildlands. University of California, Berkeley, Los Angeles

Brooks ML, D'Antonio CM, Richardson DM, Grace JB, Keeley JE, Ditomaso JM, Hobbs RJ, Pellant M, Pyke D (2004) Effects of invasive alien plants on fire regimes. BioScience 54:677–688

Bueno A, Diez JJ, Fernández MM (2010) Ophiostomatoid fungi transported by Ips sexdentatus (Coleoptera; Scolytidae) in *Pinus pinaster* in NW Spain. Silva Fenn 44:387–397

Carvalho Mendes A (2006) Implementation analysis of forest programmes: some theoretical notes and an example. Forest Policy Econ 8:512–528

Castro J, Zamora R, Hódar JA, Gómez JA (2002) Use of shrubs as nurse plants: a new technique for reforestation in Mediterranean mountains. Restor Ecol 10:297–305

Castro J, Moreno-Rueda G, Hódar JA (2009) Experimental test of post-fire management in pine forests: Impact of salvage logging versus partial cutting and non-intervention on bird-species assemblages. Conserv Biol 24:810–819

Catry FX, Bugalho M, Lopes T, Rego FC, Moreira F (2007) Post- fire effects of ungulates on the structure, abundance and diversity of the vegetation in a Mediterranean Ecosystem. In: Rokich D, Wardell-Johnson YC, Stevens J, Dixon K, McLellan R, Moss G (eds) Proceedings of the international Mediterranean ecosystems conference – Medecos XI, Kings Park and Botanic Garden, Perth, Australia

Catry FX, Bugalho M, Silva JS, Fernandes P (2010) Gestão da vegetação pós-fogo. In: Moreira F, Catry FX, Silva JS, Rego F (eds) Ecologia do Fogo e Gestão de Áreas Ardidas. ISAPress, Lisbon

Celesti-Grapow L, Alessandrini A, Arrigoni PV et al (2009) The inventory of the non-native flora of Italy. Plant Biosyst 143:386–430

Celesti-Grapow L, Alessandrini A, Arrigoni PV et al (2010) Non-native flora of Italy: species distribution and threats. Plant Biosyst 144:12–28

IPCC – Intergovernmental Panel on Climate Change (2007) Climate change 2007: impacts, adaptation, and vulnerability. Parry ML, Canziani OF, Palutikof JP, van der Linden PJ, Hanson CE (eds), Contribution of working group II to the 3rd assessment report of the intergovernmental panel on climate change Cambridge University Press, Cambridge

CMA, Conselleria de Medio Ambiente (1994) Circular de 21 de enero de 1994 sobre la extracción de madera quemada por un incendio forestal. CMA, Valencia

Cobb TP, Morissette JL, Jacobs JM, Koivula MJ, Spence JR, Largor DW (2010) Effects of post-fire salvage logging on deadwood-associated beetles. Conserv Biol 25:94–104

D'Antonio CM (2000) Fire, plant invasions, and global changes. In: Mooney HA, Hobbs RJ (eds) Invasive species in a changing world. Island Press, Washington, D.C

Daskalakou EN, Thanos CA (2004) Post-fire regeneration of Aleppo pine – the temporal pattern of seedling recruitment. Plant Ecol 171:81–89

Delitti W, Ferran A, Trabaud L, Vallejo VR (2005) Effects of fire recurrence in *Quercus coccifera* L. Shrublands of the Valencia Region (Spain): I. Plant composition and productivity. Plant Ecol 177:57–70

Dios VR, Fischer C, Colinas C (2007) Climate change effects on mediterranean forests and preventive measures. New Forest 33:29–40

Dixon KW, Roche S, Pate JS (1995) The promotive effect of smoke derived from burnt native vegetation on seed germination of Western Australian plants. Oecologia 101:185–192
Doménech R, Vila M, Pino J, Gesti J (2005) Historical land-use legacy and *Cortaderia selloana* invasion in the Mediterranean region. Glob Change Biol 11:1054–1064. doi:10.1111/j.1365-2486.2005.00965.x
Doussi MA, Thanos CA (1994) Post-fire regeneration of hardseeded plants: ecophysiology of seed germination. In: Viegas DX (ed) Proceedings of the 2nd international conference on forest fire research, Coimbra, Viegas DX Publisher, Coimbra
Espelta JM, Retana J, Habrouk A (2003a) An economic and ecological multi-criteria evaluation of reforestation methods to recover burned *Pinus nigra* forests in NE Spain. For Ecol Manag 180:185–198. doi:10.1016/S0378-1127(02)00599-6
Espelta JM, Retana J, Habrouk A (2003b) Resprouting patterns after fire and response to stool cleaning of two coexisting Mediterranean oaks with contrasting leaf habits on two different sites. For Ecol Manag 179:401–414. doi:10.1016/S0378-1127(02)00541-8
European Environmental Agency (EEA) (2007). European forest types. EEA Technical Report nr. 9. Copenhagen
FAO (2009) Global review of forest pests and diseases. FAO Forestry Paper 156, Rome
Fernández MM (2006) Colonization of fire-damaged trees by *Ips sexdentatus* (Boerner) as related to the percentage of burnt crown. Entomol Fenn 17:381–386
Fernandez Fernandez MM, Salgado CJM (1999) Susceptibility of fire-damaged pine trees (*Pinus pinaster* and *Pinus nigra*) to attacks by Ips sexdentatus and Tomicus piniperda (Coleoptera: Scolytidae). Entomol Gener 24:105–114
Ferrandis P, De las Heras J, Martínez Sanchez JJ, Herranz JM (2001) Influence of a low-intensity fire on a *Pinus halepensis* Mill. forest seed bank and its consequences on early stages of plant succession. Israel J Plant Sci 49:105–114
Flannigan MD, Stocks BJ, Wotton MB (2000) Climate change and forest fires. Sci Total Environ 262:221–229
Gayoso J, Iroumé A (1991) Compaction and soil disturbances from logging in Southern Chile. Ann Sci For 48:63–71
Gill AM, Bradstock RA (2003) Fire regimes and biodiversity: a set of postulates. Proceedings of the Australian national university fire forum, CSIRO Publishing, Melbourne, Feb 2002
Gill AM, Bradstock RA, Williams JE (2002) Fire regimes and biodiversity: legacy and vision. In: Bradstock RA, Williams JE, Gill AM (eds) Flammable Australia: the fire regimes and biodiversity of a continent. Cambridge University Press, Cambridge
Gimeno-García E, Pascual JA, Llovet J (2011) Water repellency and moisture content spatial variations under *Rosmarinus officinalis* and *Quercus coccifera* in a Mediterranean burned soil. Catena 85:48–57
Gómez-González S, Torres-Díaz C, Valencia G, Torres-Morales P, Cavieres LA, Pausas JG (2011) Anthropogenic fires increase alien and native annual species in the Chilean coastal matorral. Divers Distrib 17:58–67. doi:10.1111/j.1472-4642.2010.00728.x
Heywood VH (1995) The Mediterranean flora in the context of world biodiversity. Ecol Medit 20:11–18
Hobbs NT (2006) Large herbivores as sources of disturbance in ecosystems. In: Danell K, Bergstrom R, Duncan P, Pastor J (eds) Large herbivore ecology, ecosystem dynamics and conservation. Cambridge University Press, Cambridge
Hobbs NT, Schimel DS, Owensby CE, Ojima DS (1991) Fire and grazing in the Tallgrass Prairie: contingent effects on nitrogen budgets. Ecology 72:1374–1382
Holz S, Placci G (2005) Stimulating natural regeneration. In: Mansourian S, Vallauri D, Dudley N (eds) Forest Restoration in Landscapes. Beyond Planting Trees, Springer, New York
Hulme PE, Brundu G, Camarda I, Dalias P, Lambdon P, Lloret F, Medail F, Moragues E, Suehs C, Traveset A, Andreas Troumbis A, Vilà M (2007) Assessing the risks to Mediterranean islands ecosystems from alien plant introductions. In: Tokarska-Guzik B, Brundu G, Brock JH, Child LE, Pyšek P, Daehler C (eds) Plant invasions. Backhuys Publishers, Leiden

Izhaki I, Adar M (1997) The effects of post-fire management on bird community succession. Int J Wildland Fire 7:335–342

Jones MR, Laude HM (1960) Relations between sprouting in chamise and the physiological condition of the plant. J Range Manag 13:210–214

Kazanis D (2005) Post-fire succession in *Pinus halepensis* forests of Greece: patterns of vegetation dynamics. PhD thesis, University of Athens (in Greek with an English summary)

Kazanis D, Arianoutsou M (2004) Long-term post-fire vegetation dynamics in *Pinus halepensis* forests of central Greece: a functional-group approach. Plant Ecol 171:101–121

Keeley JE (2001) Fire and invasive species in Mediterranean-climate ecosystems of California. In: Galley P, Wilson TP (eds) Proceedings of the invasive plant workshop: the role of fire in the control and spread of invasive species. Tall Timbers Research Station, Tallahassee

Keeley JE, Bond WJ (1997) Convergent seed germination in South African fynbos and Californian chaparral. Plant Ecol 133:153–167

Keeley JE, Fotherigam CJ (2000) The role of fire in regeneration from seed. In: Fenner M (ed) Seeds: the ecology of regeneration in plant communities. CAB International, Wallingford

Keeley JE, Baer-Keeley M, Fotheringham CJ (2005) Alien plant dynamics following fire in Mediterranean-climate California shrublands. Ecol Applic 15:2109–2125. doi:10.1890/04-1222

Lamb D, Gilmour D (2003) Rehabilitation and restoration of degraded forests. IUCN, Gland, Switzerland and Cambridge, UK and WWF, Gland, Switzerland

Leone V, Logiurato A, Saracino A (1999) Serotiny in *Pinus halepensis* Mill., recent issues. In: Ne'eman G, Izhaki I (eds) Abstracts of MEDPINE, international workshop on Mediterranean pines. Beit Oren, Israel

Lindenmayer DB, Foster DR, Franklin JF, Hunter ML, Bnoss RF, Schmiegelow FA, Perry D (2004) Salvage harvesting policies after natural disturbance. Science 303:1303

Lindner M, Maroschek M, Netherer S, Kremer A, Barbati A, Garcia-Gonzalo J, Seidl R et al (2010) Climate change impacts, adaptive capacity, and vulnerability of European forest ecosystems. For Ecol Manag 259:698–709. doi:10.1016/j.foreco.2009.09.023

Liu Y, Stanturf GS (2010) Trends in global wildfire potential in a changing climate. For Ecol Manag 259:685–697. doi:10.1016/j.foreco.2009.09.002

Lloret F, López-Soria L (1993) Resprouting of *Erica multiflora* after experimental fire treatments. J Veg Sci 9:417–430

Lloret F, Vilà M (2003) Diversity patterns of plant functional types in relation to fire regime and previous land use in Mediterranean woodlands. J Veg Sci 14:387–398

Lloret F, Médail F, Brundu G, Camarda I, Moragues E, Rita J, Lambdon P, Hulme PE (2005) Species attributes and invasion success by alien plants on Mediterranean islands. J Ecol 93:512–520

Llovet J, Ruiz-Valera M, Josa R, Vallejo VR (2009) Soil responses to fire in Mediterranean forest landscapes in relation to the previous stage of land abandonment. Int J Wildland Fire 18: 222–232

López-Soria L, Castell C (1992) Comparative genet survival after fire in woody Mediterranean species. Oecologia 91:493–499

Maestre FT, Cortina J (2004) Are *Pinus halepensis* plantations useful as a restoration tool in semi-arid Mediterranean areas? For Ecol Manag 198:303–317. doi:10.1016/j.foreco.2004.05.040

Mayor AG, Bautista S, Llovet J, Bellot J (2007) Post-fire hydrological and erosional responses of a Mediterranean landscape: Seven years of catchment-scale dynamics. Catena 71:68–75

Miller G, Friedel M, Adam P, Chewings V (2010) Ecological impacts of buffel grass (*Cenchrus ciliaris* L.) invasion in central Australia – does field evidence support a fire-invasion feedback? Rangeland J 32:353–365. doi:10.1071/RJ09076

Mooney HA (1988) Lessons from Mediterranean climate regions. In: Wilson EO (ed) Biodiversity. National Academy of Sciences/Smithsonian Institution, Washington DC

Moreira F, Vallejo VR (2009) What to do after fire? post-fire restoration. In: Birot Y (ed), Living with fires, European Forest Institute Discussion Paper 15, EFI, Joensuu, Finland

Moreira F, Catry FX, Lopes T, Bugalho MN, Rego F (2009) Comparing survival and size of resprouts and planted trees for post-fire forest restoration in central Portugal. Ecol Eng 35:870–873. doi:10.1016/j.ecoleng.2008.12.017

Moreira F, Viedma O, Arianoutsou M, Curt T, Koutsias N, Rigolot E, Barbati A, Corona P, Vaz P, Xanthopoulos G, Mouillot F, Bilgili E (2011) Landscape-wildfire interactions in Southern Europe: implications for landscape management. J Env Manag 92:2389–2402
Nader G, Henkin Z, Smith E, Ingram R, Narvaez N (2007) Planned herbivory in the management of wildfire fuels. Rangelands 29:18–24
Napper (2006) BAER – Burned Area Emergency Response Treatments Catalog. USDA Forest Service. Watershed, Soil, Air Management 0625 1801 –STDTDC. San Dimas, California
Ne'eman G (1997) Regeneration of Natural Pine Forests-Review of work done after the 1989 fire in Mount Carmel, Israel. Int J Wildland Fire 7(4):295–306
Ne'eman G, Goubitz S, Nathan R (2004) Reproductive traits of *Pinus halepensis* in the light of fire—a critical review. Plant Ecol 171:69–79
Ne'eman G, Izhaki I (1998) Stability of pre- and post-fire spatial structure of pine trees in Aleppo pine forest. Ecography 21:535–542
Noble IR, Slatyer RO (1980) The use of vital attributes to predict successional changes in plant communities subject to recurrent disturbances. Vegetatio 43:5–21
Ordóñez JL, Franco S, Retana J (2004) Limitation of the recruitment of *Pinus nigra* in a gradient of post-fire environmental conditions. Ecoscience 11:296–304
Papavassiliou S, Arianoutsou M (1993) Regeneration of the leguminous herbaceous vegetation following fire in a *Pinus halepensis* forest of Attica, Greece. In: Trabaud L, Prodon R (eds) Fire in Mediterranean ecosystem, ecosystem research report no 5, Commission of the European Communities
Pausas JG (1999) Mediterranean vegetation dynamics: modelling problems and functional types. Plant Ecol 140:27–39
Pausas JG (2004) Changes in fire and climate in the eastern Iberian Peninsula (Mediterranean basin). Clim Chang 63:337–350
Pausas JG, Keeley JE (2009) A burning story: the role of fire in the history of life. BioScience 59:593–601. doi:10.1525/bio.2009.59.7.10
Pausas JG, Bladé C, Valdecantos A, Seva J, Fuentes D, Alloza J, Milagrosa A, Bautista S, Cortina J, Vallejo R (2004a) Pines and oaks in the restoration of Mediterranean landscapes of Spain: new perspectives for an old practice – a review. Plant Ecol 171:209–220
Pausas JG, Bradstock RA, Keith DA, Keeley JE, GCTE (2004b) Plant functional traits in relation to fire in crown-fire ecosystems. Ecology 85:1085–1100
Pausas JG, Keeley JE, Verdú M (2006) Inferring differential evolutionary processes of plant persistence traits in Northern Hemisphere Mediterranean fire prone ecosystems. J Ecol 94:31–39
Pausas JC, Llovet J, Rodrigo A, Vallejo R (2008) Are wildfires a disaster in the Mediterranean basin? – A review. Int J Wildland Fire 17:713–723
Pérez-Fernández MA, Rodríguez-Echeverría S (2003) Effect of smoke, charred wood, and nitrogenous compounds on seed germination of ten species from woodland in central-western Spain. J Chem Ecol 29:237–251
Peterson DL, Agee JK, Aplet GH, Dykstra DP, Graham RT, Lehmkuhl JF et al (2009) Effects of timber harvest following wildfire in western North America. USDA Forest Service, Gral. Technical Report PNW-GTR-776. Washington DC
Placci G (2005) Stimulating natural regeneration. In: Mansourian S, Vallauri D, Dudley N (eds) Forest restoration in landscapes. Beyond planting trees. Springer, New York
Poff RJ (1989) Compatibility of timber salvage operations with watershed values. In: Berg NH (Tech. coord), Proceedings of the symposium on fire and watershed management, Gen. Technical Report PSW-109. USDA, Forest Service, Pacific Southwest Forest and Range Experiment Station, Berkeley
Rego F, Rigolot E, Fernandes P, Montiel C, Silva JS (2010) Towards integrated fire management. EFI Policy Brief 4
Retana J, Espelta JM, Habrouk A, Ordóñez JL, de Solà-Morales F (2002) Regeneration patterns of three Mediterranean pines and forest changes after a large wildfire in northeastern Spain. Ecoscience 9:89–97

Robichaud PR, Beyers JL, Neary DG (2000) Evaluating the effectiveness of post-fire rehabilitation treatments. General Technical Report RMRS-GTR-63. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Fort Collins

Roy DF (1956) Salvage logging may destroy Douglas-fir reproduction. For. Res. Note 107. California Forest and Range Experiment Station

Roy J, Arianoutsou-Faraggitaki M (1985) Light quality as the environmental trigger for the germination of the post-fire species *Sarcopoterium spinosum* L. Flora 177:345–349

SAF (1996) The role of "salvage" and "sanitation" harvesting in the restoration and maintenance of of healthy forests. Society of American Foresters. www.safnet.org

Schelhaas M, Nabuurs G, Schuck A (2003) Natural disturbances in the European forests in the 19th and 20th centuries. Glob Chang Biol 9:1620–1633. doi:10.1046/j.1365-2486.2003.00684.x

Schmuck G, San-Miguel J, Camia A, Tracy D, Santos de Oliveira S, Boca R, Whitmore C, Giovando C, Liberta G, Schulte E (2010) Forest fires in Europe 2009. JRC Report 10, European Commission, Ispra, Italy

Schoennagel T, Smithwick EAH, Turner MC (2008) Landscape heterogeneity following large fires: insights from Yellowstone National Park, USA. Int J Wildland Fire 17:742–753

Scott DF, Curran MP, Robichaud PR, Wagenbrenner JW (2009) Soil erosion after forest fire. In: Cerdà A, Robichaud PR (eds) Fire effects on soils and restoration strategies. NH Science Publishers, Enfield

Simões C, Marques M (2007) The role of sprouts in the restoration of Atlantic rainforest in Southern Brazil. Restor Ecol 15:53–59

Stohlgren TJ, Barnett DT, Kartesz JT (2003) The rich get richer: patterns of plant invasions in the United States. Front Ecol Environ 1:11–14

Terry JP (1994) Soil loss from erosion plots of differing post-fire forest cover. In: Sala M, Rubio JL (eds) Soil erosion and degradation as a consequence of forest fires. Geoforma ediciones, Logrono

Thanos CA, Daskalakou EN (2000) Reproduction in *Pinus halepensis* and *P.brutia*. In: Ne'eman G, Trabaud L (eds) Ecology, Biogeography and Management of *Pinus halepensis* and *P. brutia* Forest Ecosystems in the Mediterranean Basin. Backhuys Publishers, Leiden

Thanos CA, Georghiou K (1988) Ecophysiology of fire-stimulated seed germination in *Cistus incanus* ssp. *creticus* (L.) Heywood and *Cistus salvifolius* L. Plant Cell Environ 11:841–849

Thanos CA, Rundel PW (1995) Fire-followers in chaparral: nitrogenous compounds trigger seed germination. J Ecol 83:207–216

Thompson JD (2005) Plant evolution in the Mediterranean. Oxford University Press, Oxford

Thomson VP, Leishman MR (2005) Post-fire vegetation dynamics in nutrient-enriched and non-enriched sclerophyll woodland. Austral Ecol 30:250–260

Trabaud L (1991a) Fire regimes and phytomass growth dynamics in a *Quercus coccifera* garrigue. J Veg Sci 2:307–314

Trabaud L (1991b) Is fire an agent favouring plant invasion? In: Groves RH, Di Castri F (eds) Biogeography of Mediterranean invasions. Cambridge University Press, Cambridge

Tsiouvaras C, Havlik N, Bartolome J (1989) Effects of goats on understory vegetation and fire hazard reduction in a coastal plain in California. For Sci 35:1125–1131

Valderrábano J, Torrano L (2000) The potential for using goats to control *Genista scorpius* shrubs in European black pine stands. For Ecol Manag 126:377–383

Vallejo VR (1999) Post-fire restoration in Mediterranean ecosystems. In: Eftichidis G, Balabanis P, Ghazi A (eds) Wildfire management. European Commission, Algosystems, Athens

Vallejo VR (2005) Restoring Mediterranean forests. In: Mansourian S, Vallauri D, Dudley N (eds) Forest restoration in landscapes. Beyond planting trees. Springer, New York

Vallejo VR, Alloza JA (1998) The restoration of burned lands: the case of eastern Spain. In: Moreno JM (ed) Large forest fires. Backhuys Publishers, Leiden

Vallejo VR, Aronson J, Pausas JG, Cortina J (2006) Restoration of Mediterranean woodlands. In: Van Andel J, Aronson J (eds) Restoration ecology. The new frontier. Chapter 14, Blackwell Publications, Oxford

van Wilgen BW, Richardson DM (1985) The effect of alien shrub invasions on vegetation structure and fire behaviour in South African fynbos shrublands: a simulation study. J Appl Ecol 22:955
van Wilgen BW, Forsyth GG, de Klerk H, Das S, Khuluse S, Schmitz P (2010) Fire management in Mediterranean-climate shrublands: a case study from the Cape fynbos, South Africa. J Appl Ecol 47:631–638. doi:10.1111/j.1365-2664.2010.01800.x
Vieira DL, Scariot A (2006) Principles of natural regeneration of tropical dry forests for restoration. Restor Ecol 14:11–20
Vilà M, Pino J, Font X (2007) Regional assessment of plant invasions across different habitat types. J Veg Sci 18:35–42
Wakatsuki T, Rasyidin A (1992) Rates of weathering and soil formation. Geoderma 53:251–263
Whelan RJ (1995) The ecology of fire. Cambridge studies in Ecology. Cambridge, Cambridge University
Whisenant S (2005) Managing and directing natural succession. In: Mansourian S, Vallauri D, Dudley N (eds) Forest restoration in landscapes. Beyond planting trees. Springer, New York
Wischmeier WH, Smith, DD (1978) Predicting rainfall erosion rates. A guide to conservation planning. USDA Handbook number 537. Washington, DC
Zouhar K, Smith JK, Sutherland S, Brooks ML (2008) Wildland fire in ecosystems: fire and nonnative invasive plants. General Technical Report RMRS-GTR-42-vol 6. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Ogden