Chapter 6
Effects of Climate and Extreme Events on Wildfire Regime and Their Ecological Impacts

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Abstract Fire regime has been affected by climate changes in the past, and is expected to do so in relation to the projected climate warming in the near future. For the Mediterranean Basin, higher fire risk, longer fire season, and more frequent large, severe fires are expected. The projected increased drought for the Mediterranean Basin would make ecosystems more vulnerable to fire, and more difficult to restore after fire. Ecosystem vulnerability is assessed considering soil susceptibility to post-fire erosion, and vegetation capacity to recover after fire.

In the perspective of a more severe fire regime and harsher climate, two main strategies are proposed: (1) mitigation strategies to reduce fire impacts; and (2) adaptation strategies to improve ecosystems capacity to cope with the new climate and fire regime. The focus of adaptation will be on strategies for vegetation management to reduce fire hazard, and increase ecosystem resilience, especially in highly vulnerable areas.

Restoration techniques are proposed to increase ecosystem resilience to fire by using resprouting woody species, and by increasing the diversity of species in post-fire afforestation/reforestation projects. To face increased drought, several techniques to improve water availability and water use efficiency for introduced seedling are discussed.

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Finally, the landscape dimension of fire prevention and restoration is addressed through a spatial decision support system, including a fire propagation model combined with an ecosystem vulnerability model in GIS format. The system allows assessing fire risk, identifying values at risk, and prioritizing fire prevention and post-fire restoration actions.

Keywords Fire regime • Vulnerability • Fire resilience • Plantations • Fire modeling

6.1 Climate and Fire Regime During the Last Decades in the Mediterranean Area

Fuel availability (i.e., plant biomass) and weather conditions (i.e., wind, temperature, air humidity and precipitation) are the drivers of wildfires, and both are directly or indirectly controlled by climate (Pausas 2004; Krawchuk et al. 2009). For instance, the fire regime of Mediterranean ecosystems is attributed to their seasonal climate, that is, mild temperatures and abundant rainfall in spring (which promote fuel production) followed by high temperatures and low precipitation in summer (resulting in severe water deficit, Viegas and Viegas 1994; Pausas 2004). At a longer temporal scale, changes in regional fire regimes have been associated to changes in climate through both long-term paleoecological evidence (Clark 1990) and correlation data from the twentieth century (Beer et al. 1988; Piñol et al. 1998; Westerling et al. 2006). However, neither the direct link nor the interactions between fire regime and climate are well understood because of the complexity of the underlying mechanisms. For instance, while dry conditions increase flammability and fire hazard (Piñol et al. 1998), they may also reduce plant production as well as fuel loads and continuity (Pausas and Bradstock 2007). In fact, there is currently a strong controversy with respect to the relative role of fuel and climate in the fire regime of Mediterranean ecosystems. The “fire mosaic model” proposes that catastrophic fires are due to unnatural fuel accumulations produced as a consequence of fire-suppression policies (Minnich 1983, 2001). On the other hand, the “fire weather model” states that large fires are due to extreme climatic conditions (e.g., severe drought, dry winds) and that fire is independent of the fuel type/age (Keeley et al. 1999; Keeley and Zedler 2009). Whereas in the first model, fires are fuel-limited, in the second one, fires are climate-driven. These two models have strong implications on land management; while the “fire mosaic model” suggests that fire risk may be significantly lowered by reducing fuel loads, the “fire weather model” suggests that fuel management has a limited role in reducing catastrophic fires. The large fires that occurred in the Mediterranean Basin in the last decade were related not only to extremely warm and dry weather (supporting the “fire weather model”, Founda and Giannakopoulos 2009), but also to positive anomalies in the previous wet season which promoted plant growth and fuel build-up (Trigo et al. 2006).

It has recently been suggested that fire-climate relationships are climate-dependent in such a way that the relative role of weather and fuel load varies along climatic

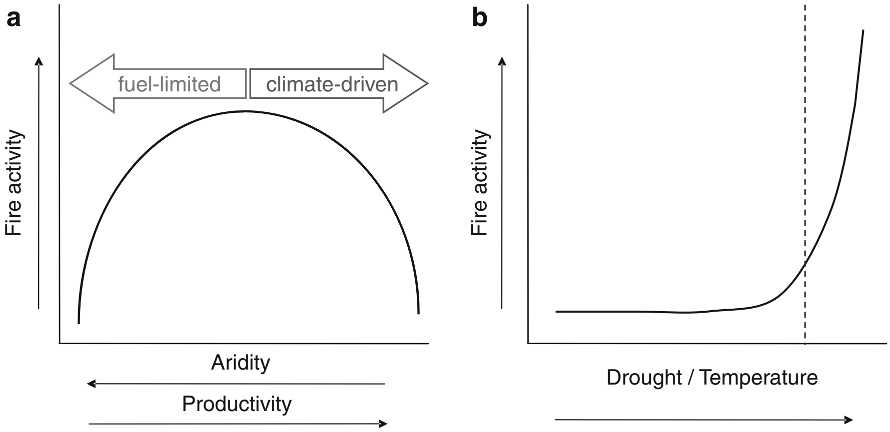

Fig. 6.1 Hypothetical fire – productivity relationship at spatial (**a**) and temporal (**b**) scale (Adapted from Pausas and Bradstock 2007)

gradients (Pausas and Bradstock 2007; Westerling and Bryant 2008; Littell et al. 2009; Fig. 6.1a). Specifically, in moist and productive ecosystems, dry conditions are needed to burn the existing fuel (vegetation), and thus the fire regime is climate-driven. On the other hand, in dry and unproductive ecosystems, fire spread is limited by both low fuel availability and low fuel continuity, even when climatic conditions are adequate for ignitions. These patterns, described for California (Westerling and Bryant 2008), seem to be applicable to the Mediterranean Basin (Pausas 2004). However, fuel load (amount and continuity) is dependent not only on climatic conditions, but also on land-use and management (Pausas and Lloret 2007). Therefore, the fire-climate relationship would be more complex in Mediterranean Basin ecosystems, where a longer and stronger human pressure has generated both fragmented landscapes and high fire ignitions (e.g., Pausas 2004).

The increase in fire activity detected in the Mediterranean Basin during the last decades has been explained by the abandonment of rural activities and the consequent fuel accumulation and increased fuel connectivity (Moreira et al. 2001; Pausas 2004; Bajocco et al. 2010). In addition, there is evidence that fire activity is linked to the climatic conditions controlling fuel availability (Viegas and Viegas 1994; Pausas 2004). All these results suggest that the fire regime in Mediterranean Basin ecosystems is globally fuel-limited (Fig. 6.1a). However, when a long time-series is considered for areas productive enough to sustain continuous fuels, a switch in the fire activity has been detected: before the 1970s, fires were small and weakly related to climate, while after this date fires were larger and strongly related to climate (Pausas and Fernández-Muñoz 2012). Before the switching point, landscapes were shaped by agriculture, livestock and other land uses, which maintained low and fragmented fuels. But the progressive land abandonment due to rural exodus to the cities resulted in burnable landscapes (i.e., fire non-limited by fuel); in such conditions, the area burnt was strongly driven by climatic conditions. Therefore, progressive changes in human land use abruptly changed from a fuel-limited fire regime to a climate-driven fire regime (Pausas and Fernández-Muñoz 2012).

The role of fuel in determining fire regimes does not depend on the amount alone, but also on its quality, type, and structure, which in turn may also be linked to climate and previous fire regimes. Therefore, fire modifies fuel amount and structure, plant composition and vegetation functioning, which in turn affect fire activity. These feedback processes can be depicted during early post-fire conditions, where lower average rooting depth may diminish plant moisture and thus increase fire susceptibility (Mouillot et al. 2002). This is especially relevant in plant communities dominated by seeders, that is, non-resprouter species whose post-fire regeneration relies exclusively on seedling recruitment (Pausas et al. 2004). At a longer time scale, flammable dry and poor communities become dominated by seeders, which in turn increase the flammability of the community (Saura-Mas et al. 2010). However, the distribution pattern of post-fire response groups along the fire gradients is still unclear because of the complexity of fire-climate spatial interactions (Clarke et al. 2005; Pausas and Bradstock 2007; Fig. 6.1a), and, thus, feedback processes are still difficult to predict.

Similarly, at the temporal scale, fire-climate interactions are not straightforward either; rather, they show a threshold effect, that is, the existence of a critical climatic value above which the probability of fire increases dramatically (Flannigan and Harrington 1988; Good et al. 2008; Westerling and Bryant 2008; Fig. 6.1b). Therefore, whereas small changes in climate conditions may seem to have little effect on ecosystem functioning, they may end up having a great impact through their effect on the fire regime, because fire may act as an amplifier of climate changes impacts. Determining the (spatial and temporal) variability of this climatic threshold is our current challenge.

6.2 Changes in Fire Regime According to Projected Climate Change in the Mediterranean

6.2.1 What Would Be New in the Forest Fire Regime?

Climate change affects variables such as air temperature, precipitation, relative humidity and wind speed, all of which influence fuel moisture and, thus, fire behavior (Moriondo et al. 2006). The fact that all the attributes describing fire regime (i.e., frequency, size, intensity, seasonality, type and severity) are highly dependent on weather and climate (Swetnam 1993; Flannigan et al. 2000) explains the rapid response that fire regimes have to changes in climate. It has even been suggested that the impacts of climate change on fire regimes might be more important than the direct impacts of climate change on species because fire can rapidly change a vegetation landscape that will fall more readily into a new equilibrium with climate (Weber and Flannigan 1997).

The inference often found in assessments of the future impact of climate change on fires is that increased drought (due to global warming) will also cause an increase in fire occurrence (Williams et al. 2001; Moreno 2005). Increases in extreme climate events, in particular, are expected to have a great impact on fire risk (Flannigan et al. 2005a). Global change has the potential to affect not only the meteorological fire hazard, but also many other interrelated components of the total fire hazard, especially its societal components (i.e., land use changes and policy, fire management strategies).

Climate-induced changes in the production of available fuel and in the overall flammability of the plant material may alter fire frequency, intensity and severity, which in turn will influence the structure and composition of ecosystems (Flannigan et al. 2000; Mouillot et al. 2002). With a climate-mediated disturbance such as fire, very complex responses to climate change may be expected (Swetnam 1993). It is likely that changes in climate will have different fire effects in different climatic conditions depending on critical thresholds of combustibility (Fig. 6.1). It has been observed, for instance, that the effect of anomalously wet years on fuel accumulation is relatively more important in dry, sparsely vegetated areas (Kipfmueller and Swetnam 2000), whereas anomalously dry conditions have a greater effect on fire danger in forested areas, where heavy fuels tend to accumulate over long periods (Agee 1993; Swetnam and Betancourt 1998; Donnegan et al. 2001).

Furthermore, changes in fire regime may have different consequences for different species (Zedler et al. 1983), and changes in species composition may have consequences on landscape combustibility and flammability, which may feed back to the fire regime. Moreover, at ecosystem level, other impacts and responses determined by plant communities as well as soil characteristics (e.g. erodibility) and post-fire events (e.g. heavy post-fire rains) may also take place.

Wildfires are already a major natural hazard in Mediterranean and other climates of the world (Westerling et al. 2006; Pausas and Keeley 2009). Wildfires include a wide range of fire regimes and affect a great diversity of ecosystem types over a large range of climates. Therefore, in principle, climate change would not introduce completely new phenomena, but it could change the trends in fire regimes and affected areas.

Studies investigating the likely effects of projected climatic changes on fire regimes began appearing around 1990 (Brown et al. 2004; Flannigan et al. 2005a), but few attempts have been made to quantify the potential impact of climate change on fire risk in ecosystems of the Mediterranean Basin (Mouillot et al. 2002; Moriondo et al. 2006). All these studies are based on a simulation approach and most of them use the outputs (climate data) obtained from General Circulation Models (GCM) run under different scenarios of fire danger indices (Brown et al. 2004; Moriondo et al. 2006).

Results generally show an increase in fire risk, burned area, fire intensity and/or frequency of fires as a result of projected climate changes (Fried et al. 2004; Moriondo et al. 2006; Flannigan et al. 2009). However, global or regional decreasing trends have also been reported (Beer and Williams 1995; Flannigan et al. 1998; Scholze et al. 2006). A decrease in fire frequency has been suggested by some studies, for instance in boreal forests (Bergeron and Flannigan 1995), indicating that the large regional variability around the world precludes any generalization about an overall increase in fire occurrence with global warming (Williams et al. 2001).

Some studies project a significant increase in fire frequency (of 40%, or even higher) under drier scenarios in relation to the reference scenario, and they suggest that these climate-change-induced modifications to fire frequency will probably be relevant for plant communities (Cary and Banks 1999). Other studies project that higher temperatures will extend the typical fire seasons, with more fires occurring earlier and later in any given year (Wotton and Flannigan 1993). The annual area burned is expected to strongly increase in some regions (Price and Rind 1994; Flannigan et al. 2005b), as are the fire danger levels (Flannigan and Van Wagner 1991; Stocks et al. 1998), the number of potential catastrophic fires (i.e. high-severity fires), and related economic losses (Fried et al. 2004). The impact of climate change differs according to vegetation fuel types, due partly to the effect of fuel type on fire intensity but mainly to the greater importance of wind speed on fire spread rate for grass fuels, as compared to brush and forest (Torn and Fried 1992). Increased fire frequency and severity could also increase the risk of losing some rare species and ecosystem types.

As for Mediterranean-type ecosystems (MTEs), fire occurrence strongly depends on the drought that drastically increases flammability during summer, on the temperature reached during this period, and on the amount of fuel load (Mouillot et al. 2002). Aiming to improve the projections of GCM-based assessments, which are somewhat hampered by coarse spatial and temporal resolutions (Stocks et al. 1998), Moriondo et al. (2006) investigated the effects of climate change on the fire risk in EU Mediterranean countries by using the output of a regional circulation model (HadRM3P) as an input to the Canadian Forest Fire Weather Index (FWI) for the current and two future IPCC scenarios (A2 and B2). Regional models are more suitable for local impact studies such as those on forest fires, especially in areas like the Mediterranean with a complex topography (Giorgi 1990). Results suggested a general increase in fire risk throughout all Mediterranean countries, with an especially strong impact likely in areas where forest land cover is high (Alps region in Italy, Pyrenees in Spain, and mountains in the Balkan region). In this study, as in others reporting similar results (Flannigan et al. 2000 for North America; Williams et al. 2001 for Australia), the higher fire risk was a direct consequence of increases in maximum temperatures and decreases in both rainfall and relative humidity during the summer period (Moriondo et al. 2006). In Spain, all GCM-based projections, under all scenarios, show a significant increase in the average monthly fire danger index, which will probably result in a lengthened fire season (Moreno 2005). Higher fire danger index values will also likely result in longer and more frequent extreme situations, even assuming that the frequency distribution of such situations remains the same, thus increasing the probability of large and severe fires. The same author suggests that impacts are expected to be higher in temperate-climate areas bordering Mediterranean ones.

Model-based climate-change assessments generally disregard various feedbacks and report a best-case forecast. Although fire-induced changes in vegetation composition, structure or distribution could create conditions that favor subsequent wildfires (Fried et al. 2004; Fulé 2008), model-based predictions do not generally consider the indirect effects of climate change on plant-growth and vegetation-distribution rates (Westman and Malanson 1992), or on community structure and composition (Ryan 1991; Mouillot et al. 2002), nor do they deal with the direct effects of increased lightning on ignitions (Price and Rind 1994). Nevertheless, despite the uncertainties in climate projections, the limitations of the modeling approaches (Pitman et al. 2007) and the complicated interacting factors, there still appears to be no reason to doubt that fire will globally increase in the coming decades (Fulé 2008; Lloret 2008).

Based on the projected climate changes described in previous chapters, we summarize below the main fire-regime changes and their impacts in Mediterranean and circum-Mediterranean countries:

- Most model-based studies tend to indicate that the projected impacts of climate change on fire regimes in Mediterranean countries (i.e., likely increases in fire frequency, intensity and severity) would have direct and significant effects on MTEs and the services they provide (Fried et al. 2004; Moriondo et al. 2006).
- Wildfires are expected to increasingly affect northern latitudes beyond the Mediterranean regions and higher elevations in mountain ranges in the Mediterranean countries. Therefore, wildfires would increasingly affect fire-sensitive ecosystems. Vulnerable forest ecosystems, such as the varied endemic Mediterranean mountain conifer forest types, could be severely endangered (Regato 2008).
- Increased land abandonment would contribute to increasing fuel load and continuity in the landscape. This would combine with more frequent extreme events to generate an increased probability of large and intense wildfires. Increased drought occurrence would also lead to higher fire frequency, especially in highly populated regions (high fire ignition probability) and the rural-urban interface.
- Both plant water stress and plant mortality will very likely increase (Rambal and Hoff 1998). Increasing amounts of decaying vegetation in drier environments are expected to enhance fuel and landscape hazardousness (Fulé 2008) by temporarily increasing dead fuels in the mid-term. This process would increase the probability of high intensity fires.
- Increasing fire recurrences will likely decrease the resilience of many MTEs (Díaz-Delgado et al. 2002; Delitti et al. 2005). In the case of forests, post-fire vegetation will less likely return to its pre-fire state because severe fires often favor alternative stable states, such as grasslands or shrublands (Fulé 2008).
- Increased drought would increase the difficulty of afforestation and reforestation in post-fire degraded lands because of the higher water stress for introduced plants.

6.3 Approaches and Methods to Identify Fire-Vulnerable Ecosystems

In the context of integrated fire management strategies aimed towards minimizing fire risk and promoting resilience to fire and biodiversity in the landscape, it seems crucial to predict how ecosystems may evolve in the short to long term after a fire.

Vulnerability has many different definitions. Based on the definition proposed by IPCC (2007), it is understood here as the degree to which a system is susceptible to, and unable to cope with, adverse effects of any driver of change (fire, in this case). Vulnerability to a given driver is a function of the character, magnitude, and rate of the driver-caused changes to which a system is exposed, its sensitivity, and its adaptive capacity. Although vulnerability aspects are frequently considered in assessment systems for most natural hazards, they have generally not been included in operational fire danger indices, which are mostly based on factors determining the fire risk conditions (Chuvieco et al. 2010).

It is known that different temporal scales may be required for analyzing different ecological processes. The post-fire soil degradation caused by water erosion, for instance, is mostly expected to occur from fire extinction up to a few months after it, when the recovery of the vegetation cover is still low (Pausas and Vallejo 1999), whereas the reestablishment of pre-fire plant communities through successional dynamics may span over several years, even decades, depending on the pre-existing vegetation types, fire severity and the environmental conditions (Keeley 2009). Therefore, the assessment of an ecosystem's vulnerability to fire should take into account the response of its various components (soil and vegetation) at different time scales.

An abundant body of literature documents the effects of fire on soils and the vegetation dynamics after fire in Mediterranean-type ecosystems, sometimes considering the interactions of fire with various other factors (Moreno and Oechel 1994; Trabaud 1994; Giovannini and Lucchesi 1997; De Luís et al. 2003; Duguy et al. 2007b; Baeza and Vallejo 2008; Duguy and Vallejo 2008). Nevertheless, integrated approaches considering the short-to-long-term response of the whole ecosystem to fire are still rare. This can be explained by the great difficulty of setting up long-term field-based experiments that would be appropriate for monitoring and documenting such responses. Given the lack of suitable field data, theoretical and modeling activities appear to be the only methods available for exploring the ecological vulnerability to forest fires in the short-to-long term for a wide range of ecosystems.

In this context, an innovative methodology aimed at assessing the ecological vulnerability to fire and based on the use of geographic information technologies (a geographic information system and remote sensing) has been developed for Mediterranean ecosystems in Spain (Duguy et al. 2012; Chuvieco et al. 2010).

This theoretical and modeling approach is structured in three stages: (1) short-term, less than 1 year after the fire, focused on soil degradation risk (Fig. 6.2); (2) medium-term, 25 years after the fire, focused on permanent changes in plant community structure and composition; and (3) integration of the short- and medium-term vulnerabilities to evaluate the overall ecological vulnerability to fire. At each stage, the variables that are considered are expressed in cartographical format and their qualitative values are successively combined by applying a matrix method. The model has been implemented in several sites: the regions of Madrid (centre Spain) Aragon (inland north-eastern Spain) and Valencia (eastern Spain). In each case, a regional-scale cartography of the ecological vulnerability to fire was obtained (Fig. 6.3). The short-term vulnerability maps facilitate the identification of the most erodible areas, which would be in greater need of short-term post-fire mitigation

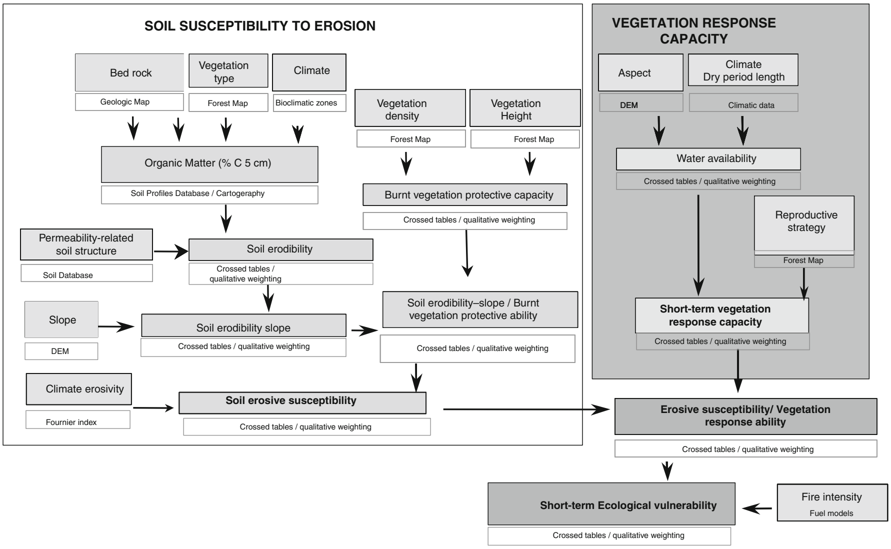

Fig. 6.2 Scheme for the short-term vulnerability analysis. Early post-fire ecosystem response depends on both physical and biotic factors, which determine post-fire soil erosion risk. The methodology considered the physical factors related to the soil susceptibility to erosion after a fire and the factors influencing the plant community response in the short term after fire. We evaluated the vegetation capacity for rapidly protecting bare soil in terms of the speed of the post-fire vegetation reestablishment

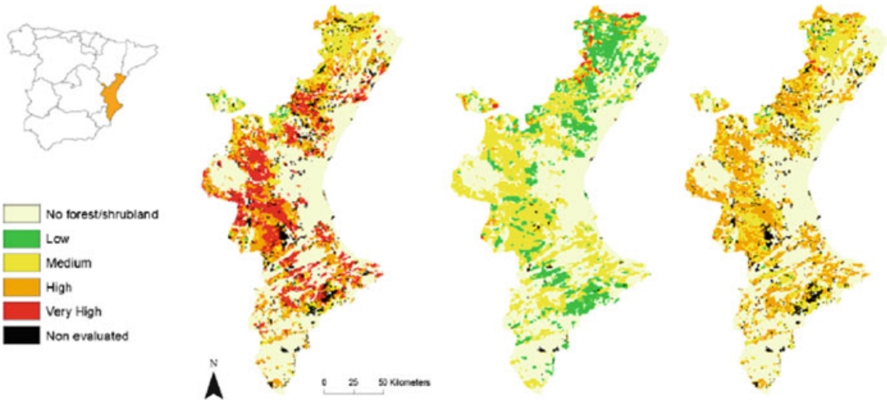

Fig. 6.3 Maps of short-term, medium-term and overall ecological vulnerability to fire (from *left to right*) for the Valencia Region (indicated in *orange* within Spain in the *upper left* localization map)

actions, whereas the medium-term vulnerability maps identify areas where permanent changes in the vegetation structure and composition can be expected in the medium term after a fire. Finally, the overall vulnerability maps indicate the most problematic situations in relation to the risk of degradation of the whole ecosystem in the medium term as a consequence of fire (Fig. 6.3).

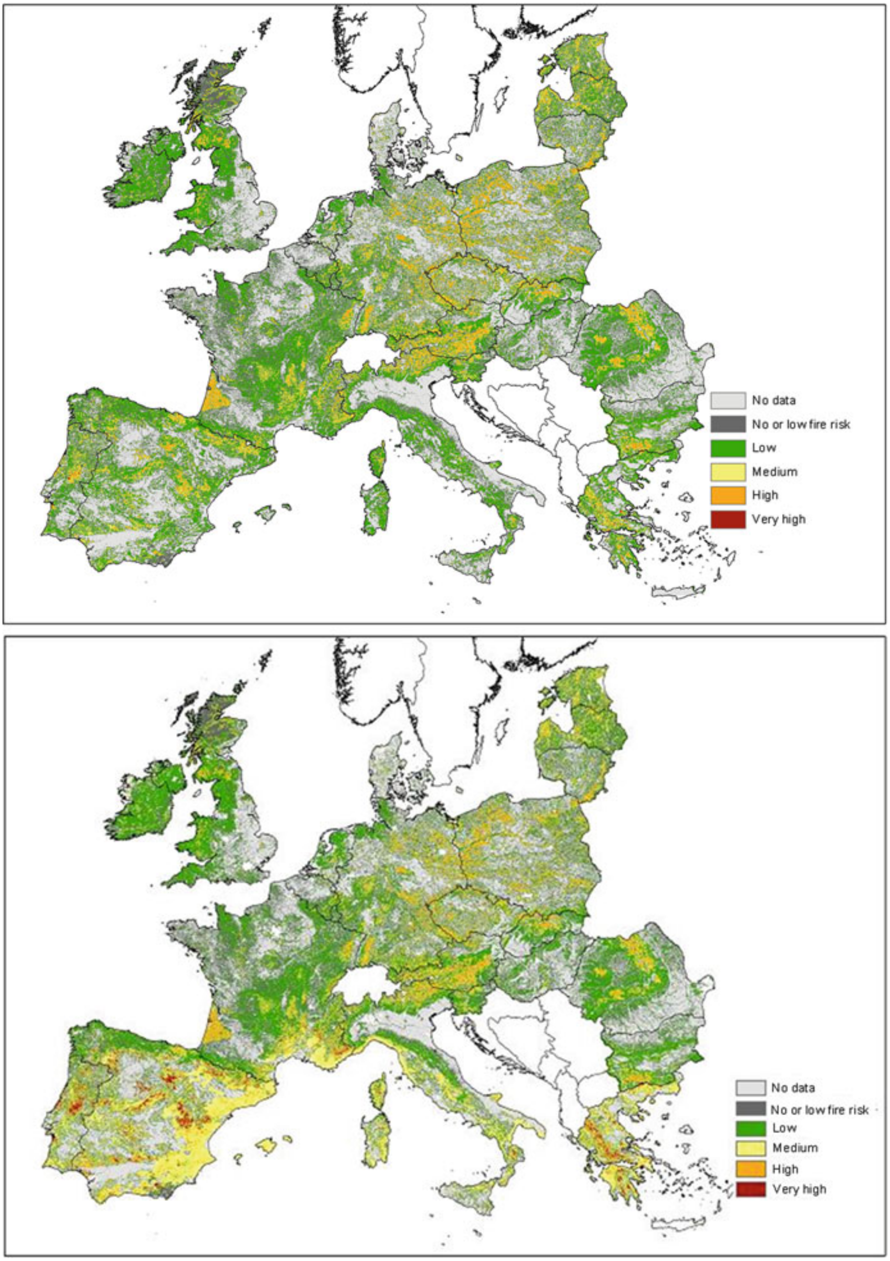

Fig. 6.4 Fire vulnerability in Europe. *Above*: Current situation. *Below*: Projection according to IPCC (2007) projections for wildfires and drought occurrence. See the text for more details

The model has been partially validated in the Ayora (Valencia) test site by comparing the overall-vulnerability values predicted by the model with field observations of medium-term post-fire ecosystem recovery. Our validation protocol used the 1979 Ayora vegetation map, corresponding to the pre-1979 fire situation (Röder et al. 2008), to run the vulnerability model. A set of previously-corrected Landsat images (Röder et al. 2008) allowed us to monitor the post-fire evolution of green biomass (NDVI) up to the year 2000, that is 21 years after the fire, in areas that had been covered by *Pinus halepensis* forests before 1979. In this way, we were able to compare the predictions of the model with real NDVI data obtained from the images. The areas of higher predicted vulnerability to fire were always associated with larger observed decreases in the green biomass (NDVI). Moreover, the model predictions were consistent for any given plant community (for each type of pine forest, in this case).

A simplified version of this approach to assessing fire vulnerability was applied to Southern and Central Europe using the CORINE2000 map for vegetation types (modified with the European Forest Genetic Resources Programme – EUFORGEN-maps) and the PESERA model (http://eusoils.jrc.ec.europa.eu/ESDB_Archive/pesera/pesera_data.html) for soil erosion risk assessment (current situation, Fig. 6.4). The regional IPCC (2007) projections on drought and forest fires risk were used to estimate changes in fire vulnerability for the end of the twenty-first century (Fig. 6.4).

6.4 Confronting Fire Impacts in Light of Climate Change

6.4.1 Post-fire Restoration Techniques to Reduce Fire Risk

In general, long-term forest fire impacts requiring restoration actions are caused by: (a) Wildfires affecting fire-sensitive ecosystems in regions where natural fires are uncommon; (b) Unprecedented fire frequency or severity – altered fire regime – over fire-dependent ecosystems; (c) Unprecedented combination of fire regime and other disturbances over fire-dependent ecosystems.

Fire impacts on ecosystems should be analyzed in terms of the interactions between direct fire-induced processes and previous human-induced degradation processes. And post-fire rehabilitation should include a long-term perspective on recuperating ecosystem integrity according to ecological restoration concepts (van Andel and Grootjans 2006). In addition, as fire hazard is inherent in Mediterranean and other ecosystems of the world, fire prevention principles should be incorporated into post-fire rehabilitation strategies in order to anticipate new fire events that will probably occur sooner or later.

In a general sense, restoration may be applied to stop ecosystem degradation after fire and to promote its regeneration. The scope of the strategies presented further on concentrates on the conservation and recovery of natural ecosystems, thus excluding from the discussion the use of exotic species or the change of land use.

In Mediterranean ecosystems affected by wildfires the main objectives of restoration programs could be (Vallejo and Alloza 1998):

1. To conserve the soil, because in terrestrial ecosystems, soil is a non-renewable primary resource which may be exposed to the risk of degradation and erosion after fire. This objective includes hydrological cycle regulation.
2. To improve ecosystem resistance and resilience in relation to fire
3. To promote mature forests, especially hardwood forests, which are scarce in Mediterranean Basin landscapes.

In the context of climate change, these objectives can be grouped into two main strategies:

(a) Mitigation strategies, which include all actions taken to reduce and reverse the impacts caused by fires (soil and water conservation), and
(b) Adaptation strategies, which encompass all approaches taken to adjust, prepare, and accommodate to the new conditions created by climate change and a new fire regime (to promote biodiverse, mature, and more resilient forests).

Mitigation techniques aim at reducing fire impacts, and adaptation strategies and methods aim at reducing fire hazard and promoting ecosystem conservation in the perspective of new fire regimes. The focus of adaptation will be on strategies for fuel and vegetation management to reduce fire occurrence and severity, and increase ecosystem resilience, especially in highly vulnerable areas.

Strategies to cope with a more severe fire regime and harsher weather conditions should address both the social and the technical components of fire management. On the social side, emphasis should be placed on improving awareness and preparedness with the aim to reduce human-caused ignitions, especially at the rural-urban interface. On the technical side, several approaches should be considered on the basis of the various threats projected with respect to fire regimes and droughts (Table 6.1).

6.4.1.1 Mitigation Strategies

Post-fire rehabilitation measures are short-term actions designed to mitigate soil degradation until natural vegetation regeneration covers the burned area. The treatments mainly aim at controlling soil erosion and runoff and preventing off-site impacts of sediments and floods. The most common post-fire rehabilitation measures are grass seeding, mulching, and contour-felled logs, as hillslope measures, and check dams (straw bales, log, and rock dams) as channel measures (Napper 2006; Cerdà and Robichaud 2009).

Emergency Seeding

Emergency seeding consists of herbaceous seeding with or without application of a mulch layer designed to promote a rapid plant cover for soil protection until the

Table 6.1 Main strategies to face a more severe fire regime and increased drought

New threats linked to changing fire and climate	Prevention	Post-fire restoration
Uncertain response of species to climate change and fire regime	Specific fire prevention measures targeted to fire-sensitive ecosystems	Increasing plant species diversity in restoration projects – application of adaptive management principles
Increased land abandonment driving increasing old field colonization by pioneer seeder plant species	Fuel control combined with the introduction of resprouting woody plant species	Introduction of resprouting woody species (see Sect. 6.4.1.2 below)
Newly affected forests	Specific fire prevention measures targeting fire-sensitive ecosystems	Reintroduction of fire-sensitive species
Increased high-intensity fire occurrence	Fuel control in the landscape to try to prevent megafires	Promote landscapes with low combustibility
Increased drought	Improve early warning at high spatial resolution for fire danger	Application of techniques to improve water inputs and water use efficiency in restoration projects

natural regeneration stabilizes the burned area. Seed mixes often include grass and legume species selected for their rapid growth rate. These mixes combine annual species to provide quick cover, and perennial species to establish longer-term protection.

The effectiveness of emergency seeding on erosion control and vegetation recovery has been widely discussed (e.g., MacDonald 1989; Beyers et al. 1998; Keeler-Wolf 1995). Robichaud et al. (2000) reviewed a number of post-fire emergency rehabilitation projects conducted in USA from 1973 to 1998, and based on the same data, Beyers (2004) discussed the effectiveness of post-fire seeding and its impacts on plant communities. According to the available data, the effect of the seeding treatment on plant cover strongly depends on both the climatic conditions and the pre-fire vegetation community.

Mulching

Together with post-fire seeding, mulching is the most widely used post-fire rehabilitation treatment, and it is mainly aimed at providing a rapid soil protection. Mulches protect the soil from rainsplash, reduce overland flow, create mini-sediment dams, reduce compaction and crusting, and increase water infiltration (Abad et al. 2000; Robichaud et al. 2000). They may also benefit plant germination and growth by changing the microclimatic conditions at the soil surface, increasing soil moisture retention (Bautista et al. 1996). Mulches mimic the role of litter. In conifer forests, low and moderate severity burned sites often have trees that are only partially consumed by fire, leaving dead needles in the canopy that fall to the ground shortly after the fire and provide a natural mulch ground cover. Pannkuk and Robichaud (2003) showed that a 50% ground cover of dead needles reduced the interrill soil erosion by 60–80%. Thus, post-fire rehabilitation treatments should exclude areas where needles provide sufficient ground cover.

The main advantages of using mulches as a post-fire emergency treatment are: they are effective immediately after installation; they reduce erosion during the critical first post-fire year; mulch materials are readily available in most areas; and thick mulch can suppress the invasive weeds that commonly appear after fires.

A study in eastern Spain (Bautista et al. 2009) tested the effectiveness of new rehabilitation treatments (seeding, mulch, and seeding plus mulch) to mitigate soil degradation and enhance vegetation recovery in the short and medium term under Mediterranean climate in burned, highly degraded woodlands. The seeded mix included several native species from different functional groups aimed not only at protecting the soil from erosion and degradation but also at enhancing ecosystem function and resilience. The species selection consisted of native herbaceous, sub-shrub and shrub species. The applied mulch was chopped wood from forest pruning activities, which mimics the effect of in-situ chopped charred wood.

The combined Seeding plus Mulching treatment enhanced total plant cover throughout the two post-fire years studied, with plant cover values being around 50% higher than for the control (untreated) plots. However, neither the Seeding nor the Mulch alone influenced the vegetation recovery.

The plant cover increase in the Seeding plus Mulch treatment was due to the germination and growth of the seeded herbaceous species. Mulch cover highly increases seed germination probably by improving soil moisture retention. The mulch layer could also play an important role in reducing seed loss downslope. Sub-shrub and shrub species also germinated and survived as part of the Seeding plus Mulching treatment, but their contribution to plant cover were low due to their relatively low percent germination and growth rate; nevertheless, as these plant species rapidly sprout after fire, surviving individuals may greatly contribute to ecosystem resilience in case of further disturbances.

In contrast to other works reporting a decrease in native species richness due to seeding (e.g., Keeley 2004) or mulching (e.g., Kruse et al. 2004), in this case there was no adverse effect of seeding or mulch treatment on the number of native species. But the Seeding plus Mulch treatment tended to decrease the total plant cover of some of the most common obligate-seeder shrub species that constitute fire-prone communities and are target species for fuel control programs (Baeza et al. 2003).

With respect to soil protection, the treatments with mulch (seeding + mulch and mulch alone) greatly reduced soil surface compaction and enhanced water infiltration. Nearly 2 years after fire and treatment application, these effects were still significant. The mulch layer also greatly reduced post-fire soil loss: the non-mulched sites showed about 20 Mg ha year-1 of soil erosion during the first post-fire year while the mulched sites had negligible losses (Bautista el al. 2009).

6.4.1.2 Fire Adaptation Measures

Adaptation strategies include all restoration actions taken to assist natural resources (species, habitats, forest, watersheds) in accommodating the new conditions imposed by climate change.

Degraded ecosystems have lost some components of the original ecosystem; some of these are evident (certain structural species: main tree and shrub species and directly associated macrofauna) but most of them are unknown or uncertain (infrequent plant species, microorganisms, etc.). In addition, degraded ecosystems have modified some functions or their rates. The main restoration strategies suggested to face the new fire regime and climate change projections are (Vallejo and Alloza 1998; Vallejo et al. 2009):

- Planting of resprouting shrubs and trees to improve adaptation (resilience and/or resistance) in fire-prone shrublands (often old fields) dominated by seeders, which are usually fuel accumulators. These areas have a high degradation risk after repeated fires. Resprouting shrubs and trees are not only very resilient to fire; they also confer resilience to the ecosystem (Ferran et al. 1991). A number of native tall shrub species considered resilient to fire were introduced with success (Vallejo 1996) in subhumid and semi-arid conditions, where they were not present, to reduce fire hazard and to improve the resilience and structure of the ecosystem.
- Combined planting of pines and hardwoods (holm oak) for mature forest restoration (Pausas et al. 2004). This is intended to combine the fast growing features of pines in degraded lands with the high resilience provided by oaks. In the forestry tradition, this operation is envisaged sequentially (Montero and Alcanda 1993): to introduce first the pines and later on the hardwoods. We have investigated the simultaneous planting of both types of species to reduce the restoration costs and to facilitate the feasibility of these operations.
- Selective clearing of highly flammable shrublands combined with the planting of resprouter species to reduce fire hazard and to improve ecosystem resilience (Valdecantos et al. 2009).

According to the proposed strategy, restoration techniques should enhance the adaptation of burned areas to both new fire regimes and climate change. The main environmental limitation for a successful introduction of plants on degraded Mediterranean sites is water stress (Vallejo et al. 2000), and this is, of course, also applicable to other arid regions of the world. In Mediterranean regions, the most critical situations are located in the transition between semi-arid and dry subhumid climates, where high water stress is combined with a high disturbance rate, especially fire.

Plantations in drylands frequently show poor results, especially when species other than pines are used. Planted seedlings often show high mortality rates, particularly when significant rainfall events are absent for more than 3 months (Alloza and Vallejo 1999). During the last decades of the twentieth century, climatic conditions in the Mediterranean basin have been exceptionally adverse, with record high global temperatures (Castro et al. 2005). Hence, drought spells may have been major drivers in the large-scale plant mortality observed. As adverse climatic conditions are likely to persist in the near future (Millan et al. 2005), current restoration techniques must be updated to make them more efficient against future climate scenarios. In addition, there is a large body of evidence indicating that a key obstacle to plantation success is transplant shock, which is the intense short-term stress experimented by seedlings as they are transferred from favorable nursery conditions to the adverse field environment (Burdett 1990).

In the long-term, newly planted forests in strategic locations might produce a positive feedback on rainfall. At regional scale, precipitation may be affected by vegetation cover. Decreasing forest cover reduces evapotranspiration and this may cause precipitation to decrease in climate conditions with high regional water circulation, for example in Moist Tropical regions – e.g., Amazonia (Correia 2006), and in coastal areas around the Mediterranean under a dominant breeze circulation regime (Millan et al. 2005). For these situations, increasing forests might increase precipitation at regional scale.

Facing Increased Drought in Plantations

The options considered to reduce water stress in plantations are (Chirino et al. 2009):

- Increase water inputs in the ecosystem (Fig. 6.5)

  Irrigation. At present, irrigation is usually conducted for plantations in arid regions at the establishment stage, but this is unusual in the Northern Mediterranean countries. Reduced and highly cost-effective irrigation systems, especially passive irrigation techniques, have been developed and applied in warm deserts all over the world (see Bainbridge 2007).

  Fog collection. Foggy areas in drylands offer the possibility of a highly valuable and inexpensive water resource which could be used for several purposes, among them for creating water points in remote areas for combating fires and for irrigating plantations (Estrela et al. 2009).

  Runoff harvesting. Runoff harvesting is an ancient passive irrigation system employed on desert hillslopes. Soil preparation involves constructing microcatchments which collect and concentrate runoff in the plantation hole (De Simón et al. 2004; Fuentes et al. 2004; Chirino et al. 2009), thus increasing seedling survival and growth.
- Increase soil water availability – soil water holding capacity

  Soil water infiltration could be improved in degraded fine-textured soils by means of mechanical soil preparation techniques (transient improvement) and by the application of mulch (Valdecantos et al. 2009).

  Both the available soil water and the soil water holding capacity can be increased by means of mechanical soil preparation and additions of organic matter

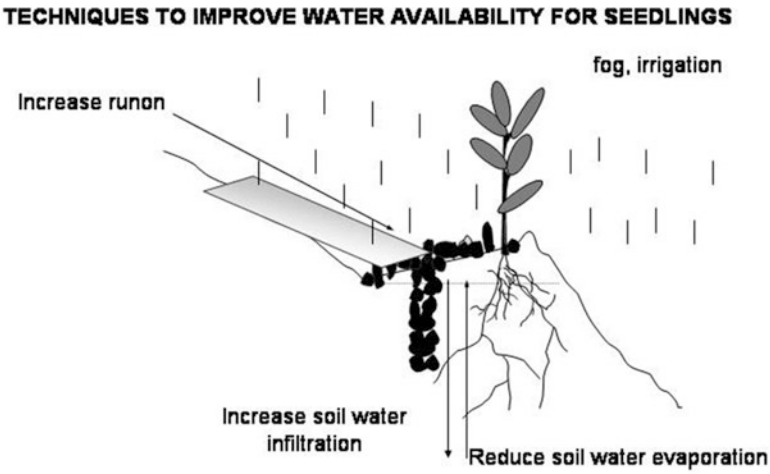

Fig. 6.5 Alternative/complementary techniques to improve water availability for seedlings in forest plantations

(e.g., biosolids, composted or uncomposted refuses: Querejeta et al. 2000), and hydrogels (Choudhary et al. 1995; Hüttermann et al. 1999). Hydrogels are probably more effective in coarse-textured soils (Seva et al. 2004). Biosolid application could have negative effects on seedling survival in relation to increased salinity and, if using semi-liquid sludges (slurry), physical problems in the soil as the sludge dries out (Valdecantos et al. 2004).

- Improvement of seedling atmospheric microhabitat: Tree-shelters

  High radiation levels and high evaporative demand characterize dry environments. Under these conditions, seedling survival is usually higher under the protection of a canopy than in open areas (Espelta 1996; Vilagrosa et al. 1997; Vallejo et al. 2006), but exceptions are not uncommon (Vilagrosa et al. 2001; Pérez-Devesa et al. 2004). The use of tree shelters may ameliorate harsh conditions and improve species survival and growth. These positive effects have been attributed to the fact that tree shelters modify the plant environment: they create a greenhouse microclimate with increases in temperature, relative humidity, and carbon dioxide levels (Burger et al. 1992).

  Most of the species tested under Mediterranean conditions showed a positive response to tree shelters (Costello et al. 1996), with the effect of tree shelters being more relevant, in terms of survival, in the driest regions.

- Improvement in seedling water use efficiency

  – Plant species selection
  – Plant quality manipulation

Plant Species Selection

Restoration usually consists of introducing one or several keystone species. These species, typically trees or tall shrubs, are supposed to play a critical role in determining ecosystem structure and functioning, acting as 'ecosystem engineers' (Jones et al. 1994) which are able to modify the habitat. It is assumed that these species would improve soil properties, create a forest floor habitat, improve the microclimate, indirectly facilitate the importation of seeds by birds and so on. Finally, the introduction of a woody species would not be enough for its complete establishment if symbionts, pollinators or dispersers were lacking (Hobbs and Norton 1996). Mycorrhiza and/or rhizobacteria inoculation in the nursery is a way to ensure efficient symbiosis for seedlings to be introduced (Barea and Honrubia 2004).

Within the set of native species found to be suitable for restoring a given habitat, we have to select the ones that best fit the management objectives proposed. In the case of post-fire restoration we would select woody resprouters according to the above-stated objectives of increasing fire ecosystem resilience and reducing fire risk. Moreover, given the projection of increased drought we would prioritize the use of drought-resistant native woody resprouter species, ecotypes, and genotypes, especially for very degraded sites and dry microclimates.

Plant Quality: Nursery Cultivation

Suitable restoration techniques may help the seedlings to get through the transplant shock and the first summer drought, and thus establish successfully. These include several nursery techniques that take into account the morpho-functional characteristics of seedlings to promote their resistance to drought and increase their acclimation to the reforestation site.

The main technical elements in the nursery culture are:

- Substrates or growing media.
- Containers.
- Drought preconditioning.

Substrates or Growing Media

The characteristics of the growing media are important for good root development, which is considered a key step in the success of a plantation (Peñuelas and Ocaña 1996). Nowadays, the growing media recommended for use include standard components like peat moss or other alternative organic materials such as coconut fiber, composted

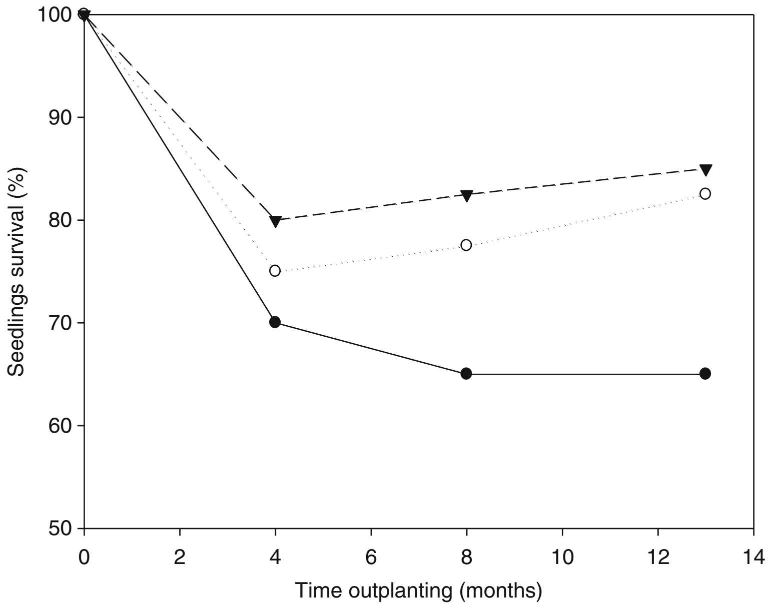

Fig. 6.6 Survival of *Quercus suber* seedlings after 13 months in outplanting (Control substrate: *black circle* and *solid line*; hydrogel stockosorb – 0.7%: *white circle* and *dotted line*; hydrogel stockosorb – 1.5%: *black triangle* and *long dashed line*) (Chirino et al. 2009)

sawdust, bark, or composted sewage sludge in combination with a mixture of aeration materials like perlite, sand, vermiculite, tuff or polystyrene (Landis et al. 1990).

A mixture with low proportions of other substances like hydrogels or some clays (sepiolite) can increase the water holding capacity of the plug, thus providing the seedlings with higher water availability for a longer period of time in the field. This fact can be especially important in climates with high rainfall variability, like the semi-arid climate. Field results ratified the beneficial effects that mixing hydrogels into the substrate had on seedling performance (Fig. 6.6).

Containers and Root Systems

Several studies have related the planting stock quality of the seedlings to the type of container used (Landis et al. 1990; Peñuelas 1995; Vilagrosa et al. 1997; Dominguez et al. 1999). An appropriate container should have a shape and dimension that allow the seedling to develop correctly, especially its root system. In general, high-volume containers (300 cm^3 or more) are recommended for reforestations in dry and semi-arid climates and for species with high root-to-shoot ratio, because they enable the good root system development that is critical during the first stages after plantation. According to our experience, long containers are preferred for species that develop a tap root, like *Quercus* sp., whilst wider containers are recommended for species that show important secondary-root development.

Drought Preconditioning

Drought preconditioning is one of the main techniques used to precondition seedlings against drought stress by means of induction of mechanisms for drought resistance. However, because Mediterranean plant species have ontogenetically high resistance to stress conditions, the common techniques of drought preconditioning (i.e., short-term preconditioning) applied to species characteristic of humid or subhumid climates are not very effective when applied in Mediterranean dry or semiarid species (Fonseca 1999; Vilagrosa et al. 2003). Experiments carried out by CEAM showed that long-term drought preconditioning in the nursery promotes higher benefits to plant morpho-functional characteristics than short-term preconditioning (Rubio et al. 2001; Chirino et al. 2003). On the other hand, the response of species to drought preconditioning seems to depend on the plant species. For example, species like *Pistacia lentiscus* are very responsive to preconditioning whilst species like *Quercus coccifera* are not. Probably, this type of response is related to the drought strategies developed for each species (Vilagrosa et al. 2003).

The main responses obtained in drought preconditioning experiments are higher root-shoot ratio in the nursery (Chirino et al. 2003), changes in allocation patterns (i.e., higher fine-root colonization in the plantation hole and lower above-ground development) (Rubio et al. 2001; Chirino et al. 2003), higher tolerance to drought conditions through higher elasticity of cell membranes (Rubio et al. 2001) or better photochemical efficiency (Vilagrosa et al. 2003), drought-avoidance mechanisms like higher root hydraulic conductivity to supply water to leaves, higher leaf capacitance to water and lower transpiration rates (Villar-Salvador et al. 1999; Vilagrosa et al. 2003). Although drought preconditioning does not improve survival, seedlings are generally better adapted to field conditions (Rubio et al. 2001). For example, preconditioned seedlings of *Q. suber* and *P. lentiscus* showed lower biomass reduction due to summer drought than well irrigated seedlings.

6.4.2 Landscape Dimension in Fire Prevention and Restoration

Efficient and sustainable fire management policies (and particularly those related to fire prevention and post-fire restoration) need to be planned at regional and landscape levels (Fernandes 2006; Finney 2007; Schmidt et al. 2008), and their effects should also be evaluated at wider spatial and temporal scales than those resulting from single fire events (Lloret 2008). Such upscaling-based approaches may also allow better analyses of the interactions between wildfires and the environmental changes that take place at a global scale.

One of the main questions that current fire-related research needs to address is how to manage fire-prone landscapes under climate change in order to reduce both the future fire risk and the vulnerability of landscapes to fire (i.e., to increase their resilience to fire). The hypothesis underlying this research is that appropriate landscape-level fuel management (resulting in the long-term modification of the structure, composition and spatial configuration of plant communities) could alter both the ecosystem-level successional trajectories and the landscape structure, facilitating a dynamics towards more resistant (less flammable) and resilient ecosystems and landscapes.

It is broadly accepted that not only the nature of fuels and their moisture content, but also their spatial distribution in the landscape, among other factors, have a strong influence on fire spread and behavior (Turner and Romme 1994; Mouillot 2001; Duguy et al. 2007a, b), and, thus, on potential fire impacts. Higher degrees of landscape fragmentation (i.e., a more fine-grained landscape) have often been observed to limit fire propagation and moderate fire behavior (Minnich 1983; Knight 1987; Duguy et al. 2007a, b). Traditionally, it is believed that disturbances are more likely to spread across a homogeneous area (Wiens et al. 1985), but the opposite also occurs (Turner 1987). It has likewise been proposed that, in highly fragmented landscapes, disturbances require a higher boundary-crossing frequency and a more convoluted route and, therefore, spread less easily (Turner and Romme 1994; Forman 1995). In the case of fires, it is generally accepted that greater landscape heterogeneity retards fire propagation (Minnich 1983; Wiens et al. 1985; Knight 1987), although landscape pattern may have little influence on crown fire behavior when burning conditions are extreme (Turner et al. 1994: Keeley et al. 1999), which might become more frequent in the future. Indeed, no universal correlation has been found between fire propagation rate and landscape heterogeneity (Morvan et al. 1995).

In the last decades, the intense land abandonment that took place in most northern Mediterranean landscapes generally caused the disappearance of the former mosaic-like landscape structure, conformed by small patches of natural vegetation in the matrix of agricultural-dominated land (Lloret et al. 2002; Duguy 2003). The increase in the continuity of natural vegetation patches led to a loss of landscape heterogeneity and fragmentation (Lloret et al. 2002), which, in turn, favored the spread of large and intense fires which have often resulted in a further homogenization of these landscapes (Debussche et al. 1987; Vos 1993; Vázquez and Moreno 1998; Lloret et al. 2002). These trends will likely be maintained, or even enhanced, under future conditions (Moreno 2009), and interconnected highly-flammable patches of increasing size might result in a strong increase in the risk of large fires.

An effective fuel management policy in relation to fire control requires the development of models and procedures that optimize its effectiveness, both spatially and temporally, and that minimize the arbitrariness in its planning process (Hiers et al. 2003; Fernandes 2006). The temporal efficiency should aim to extend the interval between treatments (Baeza and Duguy 2009), whereas the spatial efficiency should aim to minimize the ratio of area treated in the landscape to the expected benefits (i.e., decrease the risk of large fires) (Loehle 2004).

Fire management has to be assessed through costs, benefits and damages in the long term (Pausas and Vallejo 2008). Given the potentially high costs of landscape-level fuel treatments for fire control or for enhancing fire resilience, it is essential to plan these actions through spatial optimization procedures (Hiers et al. 2003; Finney 2004). In the context of climate change, it will be particularly crucial to apply cost/benefit analyses to optimize the resources needed for fire mitigation and restoration actions (Martell 2001; Moreno 2005).

Forman and Collinge (1996) described the "aggregate-with-outliers" model as an effective landscape structure in relation to fire spread control and biodiversity enhancement. To reduce both fire occurrence and fire spread, while promoting the expansion of forest in the landscape, these authors proposed three main approaches focused on landscape pattern:

1. Minimize the sites that are especially susceptible to fire ignition
2. Increase landscape spatial heterogeneity
3. Increase barriers or filters that inhibit fire spread.

Given the difficulty and obvious limitations in implementing large-scale and long-term experiments on fuel treatment, and in assessing their performance in relation to real fires, our indications as to the effectiveness of this "*aggregate-with-outliers*" model, or of any other proposed landscape structure in relation to fire control, mostly come from recent theoretical and modeling studies (Hirsch et al. 2001; Hiers et al. 2003; Finney 2004; Finney et al. 2007). Many questions related to fuel treatments, such as their optimized placement in the landscape or their potential effectiveness under alternative climate change scenarios, can only be addressed through modeling approaches (Finney 2001b).

Most modeling-based studies confirm that fuel treatments need to be designed and implemented at a landscape level in order to significantly modify the spatial pattern of fire spread and behavior (Finney 2001a, b; Stratton 2004; LaCroix et al. 2006; Duguy et al. 2007a, b). A strategic placement of theoretical treatments in the landscape results in a more effective reduction of fire propagation and a more moderate fire behavior than a random or arbitrary arrangement of treatments (Finney 2001a, 2003, 2007; Loehle 2004; Duguy et al. 2007a, b; Schmidt et al. 2008). The overall efficiency of the action is also improved, i.e. we obtain a larger ratio of area saved (non-burned) to total area treated in the landscape (Loehle 2004).

Real landscapes are characterized, however, by their complexity and fine-scale variability in terms of fuels, topography, and weather, which produce complex patterns of fire behavior and effects (Finney 2004). Analytical solutions to the optimization of fuel treatment placement on real landscapes are still under investigation.

The combination of spatial technologies (GIS) and fire modeling, and their integration with ecological principles, multi-criteria decision methods and other models (vegetation, landscape, watershed, treatments) have led to the development of spatial decision support systems (SDSS) to aid in the complex multi-objective process of forest management in fire-prone ecosystems (Hiers et al. 2003; Sisk et al. 2006). SDSSs can help land managers in designing dynamic and sustainable landscape-specific prevention and restoration plans under different scenarios of global change. We present here an innovative GIS-based procedure for fuel treatment optimization in the landscape, which we developed in the framework of the SDSS ForestERA (http://www.forestera.nau.edu/). We implemented the spatial procedure in a test area (Ayora, Valencia) for which we had previously parameterized the FARSITE model (Duguy et al. 2007a, b) in relation to a set of management objectives (i.e., minimization of fire risk, promotion of landscape resilience to fire, promotion of biodiversity, conservation of soil and water). We then carried out a preliminary exploration of the effectiveness of various fuel scenarios for controlling fire propagation and moderating fire behavior (Duguy et al. 2009).

We first carried out an assessment of the risks that the studied landscape might face from the threat of catastrophic wildfire and its consequences. We identified Fire Hazard (assessed through the variable Heat per unit of area, in kJ m^{-2}), Crown Fire Behaviour and Ecological Vulnerability to Fire (which included post-fire erosion potential) as the three key risk factors. The latter variable was assessed through a model of ecological vulnerability to forest fires in Mediterranean ecosystems (Duguy et al. 2012), as explained in Sect. 6.3. These three layers were combined in ArcGIS to create a composite risk layer (the layer *Risks*, Fig. 6.7). We then identified and prioritized features or areas of particularly high importance in the landscape, i.e., areas in critical need of protection from catastrophic wildfires (Fig. 6.8a). We combined this information with the previous assessment of the risks through overlay and buffering processes to create the final layer *Values* (Fig. 6.8b). A final overlay was generated through the combination of the *Values* and *Risks* layers. The various resulting combinations were evaluated through an expert knowledge-based decision table, leading to the final prioritization map (Fig. 6.9). We finally created various fuel scenarios aiming to minimize fire hazard through the reduction of fuel loads in some of the areas that the previous analysis had identified as most in need of management attention. We simulated both extensive hazardous fuel removals (fire prevention treatment) and the introduction of wooded patches in different successional stages (fire restoration action). We modeled fire behavior with FlamMap (Finney 2006) and FARSITE (Finney 1998) in all scenarios and compared them using four outputs from those programs: the rate of fire spread ($m.min^{-1}$), the fireline intensity (kW m^{-1}), the heat per unit of area (kJ m^{-2}), and the crown fire activity.

Preliminary results confirm that fuel spatial distribution is a key parameter influencing fire propagation and behavior across the landscape. Minor or moderate changes in the spatial pattern of fuels may cause substantial changes in fire behavior.

In the studied landscape, concentrating fuel reduction treatments on heavy surface fuel types, such as fuel model 4-type shrublands, which favor the spread of intense fires (Anderson 1982), allowed us to substantially moderate fire behavior and, thus, to reduce potential fire-caused damages to the aboveground vegetation and to the whole ecosystem.

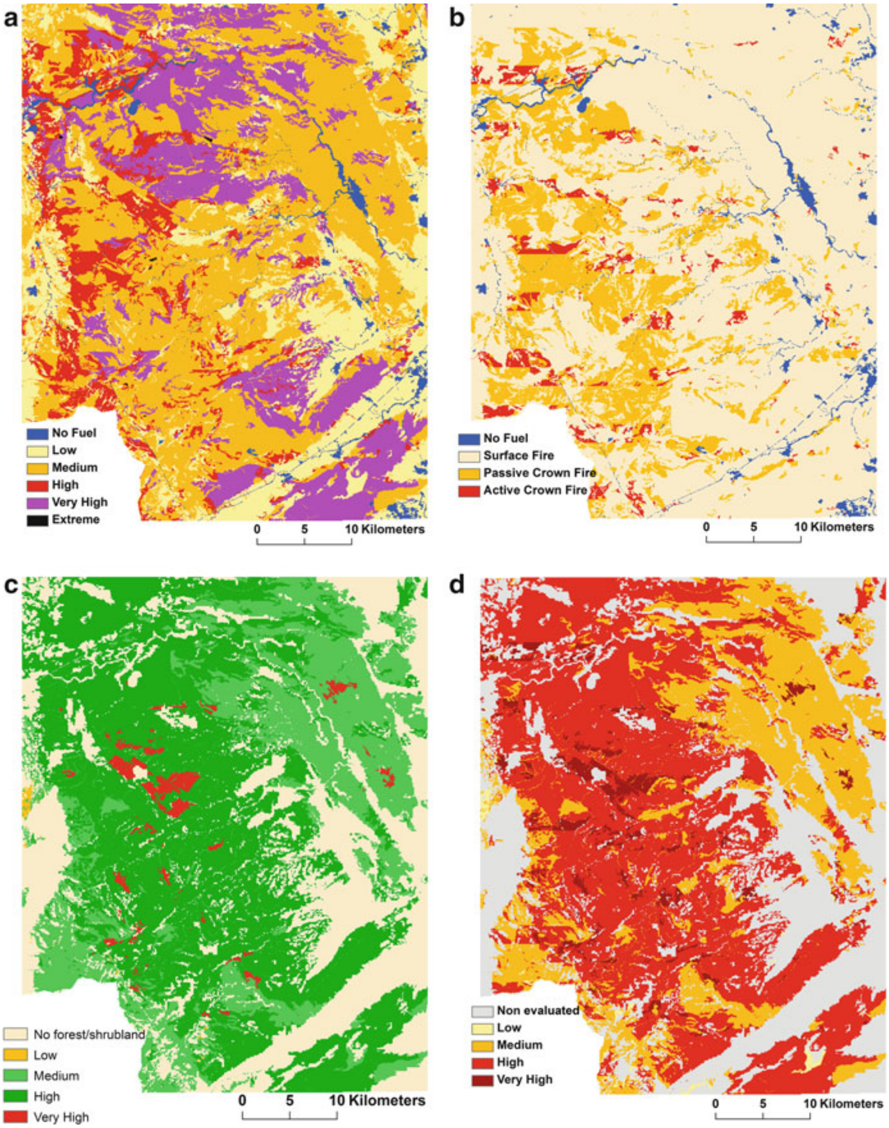

Fig. 6.7 Composite Risk layer (**d**) developed by overlaying Fire Hazard, or predicted Heat per Unit Area in kJ m^{-2} (**a**), Crown Fire Behaviour (**b**) and Ecological Vulnerability (**c**) layers. In (**a**): Low (<3,000); Medium (3,000–10,000); High (10,000–20,000); Very high (20,000–32,000); Extreme (≥32,000) kJ m^{-2}

Simulations confirmed that the creation of a more fine-grained landscape through the fragmentation of large fire-prone areas (fuel model 4) with woodlands in different successional stages could be very effective for reducing fire size and, in most cases, burning conditions. However, habitat fragmentation may have negative effects on biodiversity; therefore both fire prevention and biodiversity conservation should

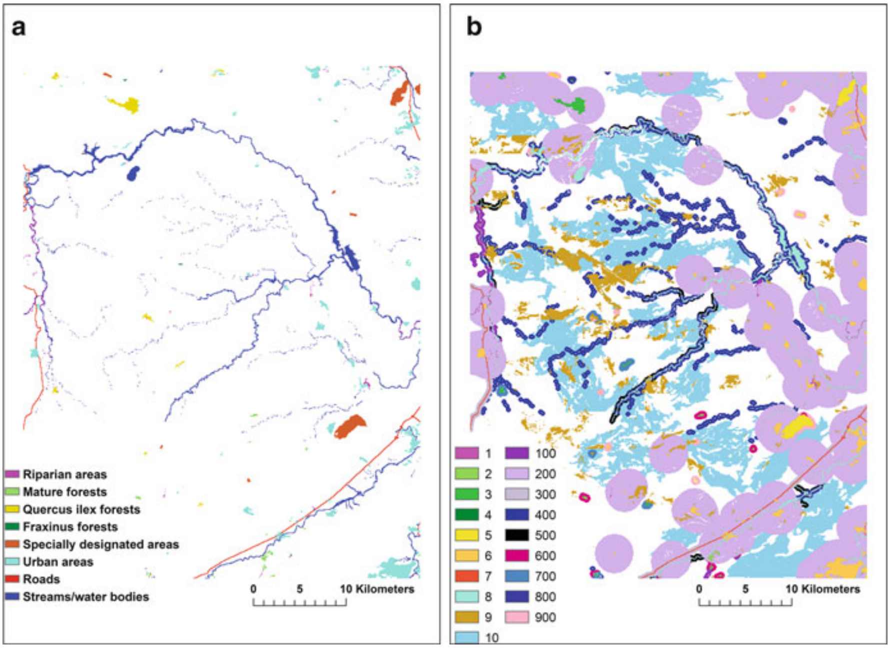

Fig. 6.8 Critical features in need of protection from wildfire (**a**) and Areas of importance generated through overlay/buffering processes (**b**). In (**b**), we include critical features (from (**a**)) and their buffers, adding all areas at very high risk and the largest continuous areas (>1,000 ha) at high risk, after the composite risk map (categories 9 and 10, respectively). The first 8 legend numbers in (**a**) correspond to those in (**b**). Buffered areas in (**b**): 100: Riparian; 200: Urban; 300: Road; 400: Water body; 500: Stream; 600: Mature forest; 700: *Quercus ilex* forest; 800: *Fraxinus* forest; 900: Specially designated areas

be integrated in the planning process. In the absence of fire, the landscape structure that would result from these actions would probably enhance the extension of woodlands in the medium-to-long term, thus promoting biodiversity.

6.5 Concluding Remarks

Climate change is expected to trigger a more severe fire regime and more difficult conditions for ecosystem restoration after fire. At present, strategies and techniques are available to address the long-term ecological restoration of degraded ecosystems/landscapes after wildfires. Nevertheless, the restoration process is subject to many uncertainties as the restorationist cannot foresee all the possible environmental circumstances that might affect the success of a restoration, nor is all the knowledge available on the ecosystem to be restored or on the potential multiple interactions between introduced plants, soil properties, extant organisms and so on. Climate Change introduces new uncertainties both

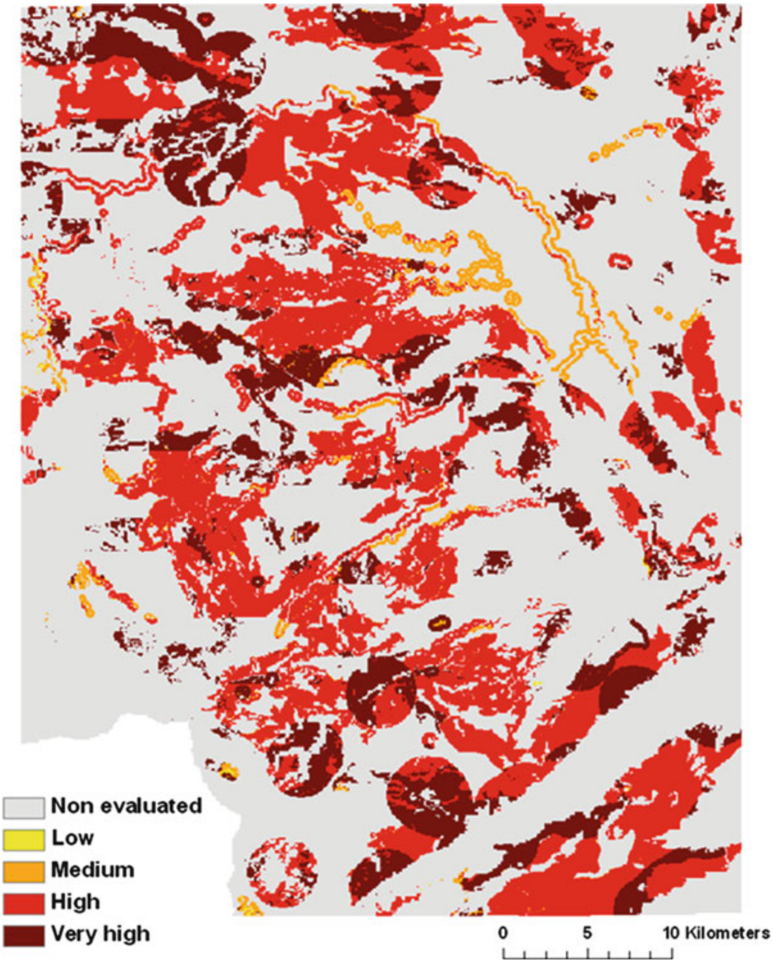

Fig. 6.9 Priority areas for management attention. Non fuel areas (urban, roads, streams and water bodies) were excluded as well as specially designated areas

in its magnitude and in its consequences on wildfires and organisms. Therefore, restoration projects should follow adaptive management principles (Whisenant 1999), including monitoring and the possibility of rectifying or amending the restoration actions as we learn from the dynamics of the restored land. The shortcoming of this approach is that it requires further and longer-term funding.

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