

# Chapter 11

## Management of Threatened, High Conservation Value, Forest Hotspots Under Changing Fire Regimes

Margarita Arianoutsou, Vittorio Leone, Daniel Moya, Raffaella Lovreglio, Pinelopi Delipetrou, and Jorge de las Heras

### 11.1 The Biodiversity Hotspots of the Earth

Biodiversity hotspots are geographic areas that have high levels of species diversity but significant habitat loss. The term was coined by Norman Myers to indicate areas of the globe which should be a conservation priority (Myers 1988).

A biodiversity hotspot can therefore be defined as a region with a high proportion of endemic species that has already lost a significant part of its geographic original extent. Each hotspot is a biogeographic unit and features specific biota or communities. The current tally includes 34 hotspots (Fig. 11.1) where over half of the plant species and 42% of terrestrial vertebrate species are endemic. Such hotspots account for more than 60% of the world's known plant, bird, mammal, reptile, and amphibian

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M. Arianoutsou (✉)

Department of Ecology and Systematics, Faculty of Biology, School of Sciences,  
National and Kapodistrian University of Athens, Athens, Greece  
e-mail: marianou@biol.uoa.gr

V. Leone

Faculty of Agriculture, University of Basilicata, Potenza, Italy  
e-mail: vittorio.leone@tiscali.it

D. Moya • J. de las Heras

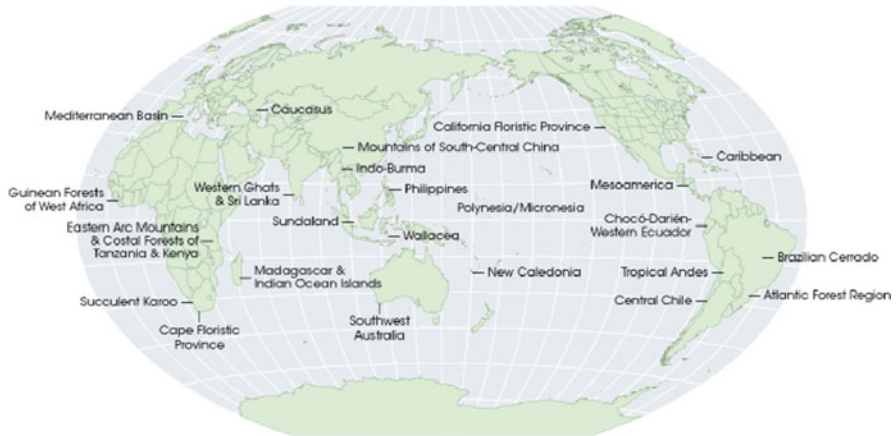
ETSI Agronomos, University of Castilla-La Mancha, Albacete, Spain  
e-mail: daniel.moya@uclm.es; Jorge.heras@uclm.es

R. Lovreglio

Faculty of Agriculture, University of Sassari, Sardinia, Italy  
e-mail: rlovreglio@uniss.it

P. Delipetrou

Department of Botany, Faculty of Biology, School of Sciences,  
National and Kapodistrian University of Athens, Athens, Greece  
e-mail: pindel@biol.uoa.gr



**Fig. 11.1** Global Biodiversity hotspots distribution. (Source: <http://earthobservatory.nasa.gov/Features/Conservation>)

species whose distribution area was originally restricted to 15.5% of the Earth's land surface and has now shrunk to 2.2%.

The principles of biodiversity hotspots have been the basis for conservation efforts by Conservation International and other leading environmental groups to distinguish a global set of high-priority terrestrial ecoregions for conservation.

### ***11.1.1 The Biodiversity of the Mediterranean Basin Hotspot***

The Mediterranean Basin is one of the 34 biodiversity hotspots included in the current list of Conservation International (2005). It encompasses North Africa (Morocco, Algeria, Tunisia, Libya, Egypt, Israel, Lebanon, Jordan and Syria), Turkey, Greece, Malta, Cyprus, all the Adriatic coastal countries (Montenegro, Albania, Croatia), almost all Italy and major islands, Mediterranean France (PACA Region and Corsica,) Spain and Portugal, including the islands in the Atlantic Ocean which pertain to the last two countries [the Macaronesian Islands of the Canaries, Madeira, the Selvagens, the Azores, and Cape Verde]. Of the 22,500 species of vascular plants in this hotspot, approximately 11,700 (52%) are found nowhere else in the world (Greuter 1991, 1994; Médail and Quézel 1999).

Having about 2,085,292 km<sup>2</sup> of surface, the Mediterranean Basin is located at the intersection of two major landmasses, Eurasia and Africa, which contributes to its high diversity and spectacular scenery, made of thousands of islands, abrupt coasts and sandy beaches, high elevation mountains up to 4,808 m (Monte Bianco, Italy), a rugged topography and a wide variety of climatic types. The broad climatic type prevailing over the Mediterranean Basin could be defined as a Dry Summer Temperate climate under the Köppen climate classification.

Although much of the Mediterranean Basin was thought to be once covered by evergreen and deciduous oak forests as well as conifer forests, 8,000 years of human

settlement and habitat modification have distinctly altered the characteristic vegetation (Quézel 1985; Zohary 1973). Today, the most widespread vegetation type is hard-leaved or sclerophyllous shrublands called *maquis* or *matorral*, which include representatives from the plant genera *Juniperus*, *Myrtus*, *Olea*, *Phillyrea*, *Pistacia*, and *Quercus*. Some important components of Mediterranean vegetation (species of the genera *Arbutus*, *Calluna*, *Ceratonia*, *Chamaerops*, and *Laurus*) are tropical relicts from the ancient forests that dominated the Basin two million years ago. Frequent burning of maquis (mainly for grazing purposes) resulted in depauperate vegetation dominated by Kermes oak (*Quercus coccifera*), *Cistus* spp., *Sarcopoterium spinosum* or *Genista fasselata*, all of which regenerate rapidly after fire by resprouting or mass seed germination. Shrublands, including maquis and the aromatic, soft-leaved and seldom drought deciduous phrygana of *Rosmarinus*, *Salvia*, *Thymus* or *Coridothymus* persist in the semi-arid, lowland, and coastal regions.

The Mediterranean Region harbours a high degree of tree richness and endemism (290 indigenous tree species of which 201 endemics). A number of trees are important flagships, including the cedars (such as the famous cedar of Lebanon, *Cedrus libani*, which has been exploited since the rise of civilization in the Fertile Crescent); the argan tree (*Argania spinosa*), a species in the Souss region of southwest Morocco; oriental sweet gum (*Liquidambar orientalis*); and Cretan date palm (*Phoenix theophrastii*) in Crete (Greece) and western Turkey.

Many of the hotspots occur in regions with high levels of population density, a potential factor of threat for their conservation status and overall existence. Population trends in many biodiversity hotspots indicate a high risk of ongoing habitat degradation (Cincotta et al. 2000).

The greatest impacts of human civilization in the Mediterranean have been deforestation, intensive grazing and fires, and urbanization, especially along the coast. At present, Mediterranean forests cover about 9.4% of the region's total land area (Leone and Lovreglio 2004). The Mediterranean region is now facing a turbulent transformation of its political, human and settlement patterns. Dramatic changes in population density and distribution mark the area, which includes one of the most over-crowded coastlines in the world.

In terms of habitat deterioration, population density is, among others, strongly related with forest fires: a number of papers relate number of fires to population density with more or less robust relationships (Cardille et al. 2001; Leone et al. 2003; Syphard et al. 2007; Martínez et al. 2009). Among other factors threatening biodiversity hotspots fires represent one of the most detrimental.

## 11.2 Climate Change and Fire Regime Interaction Affect New Areas

Global warming represents perhaps the most pervasive of the various threats to the planet's biodiversity, given its potential to affect even areas far from human settlements. Recent reports outline the extensive biological changes that are ongoing

because of global warming (Parmesan and Yohe 2003; Root et al. 2003). Malcolm et al. (2006) highlighted the potential impacts of global warming on biodiversity hotspots. Projected extinctions in hotspots under doubled CO<sub>2</sub> climates represented 39–43% of the biota, some 56,000 endemic plant species and 3,700 endemic vertebrate species. For specific hotspots, projections predicted extinctions of more than 3,000 plant species (Cape Floristic Region, Caribbean, Mediterranean Basin and Tropical Andes). For the Mediterranean Basin air temperature is foreseen to increase between 2% and 4% over the next century (Palutikof and Wigley 1996), while precipitation is predicted to decrease in autumn and increase in winter (Deque et al. 1998). Models predict changes in frequency, intensity and duration of extreme events, with more hot days, heat waves and heavy precipitation events, and fewer cold days (Lindner et al. 2008). The main expected impacts of climatic change on Mediterranean forests, namely on stability of forest ecosystems and natural disturbances (e.g. fire, pests, wind-storms) among others are strongly negative (Alcamo et al. 2007; Rosenzweig et al. 2007) and they are expected to affect plant life (e.g. growth, litter production), species recruitment, community composition and biodiversity, and regeneration processes after fire (Lavorel et al. 1998).

Climate warming is likely to have a rapid and profound impact on fire activity in several vegetation zones. For the Mediterranean Europe, Piñol et al. (1998) have studied a climatic series of 50 years from a locality in southern Spain and two fire hazard indices, and concluded that an effect of climate warming on wildfire occurrence is evident. Pausas (2004) analysed data from 350 meteorological stations in the eastern Iberian Peninsula covering a time period of 50 years (1950–2000) and fire records for the same area. He concluded that a clear pattern of increasing number of fires and size of area burned during the last century is observed, related to increasing mean annual as well as summer temperatures.

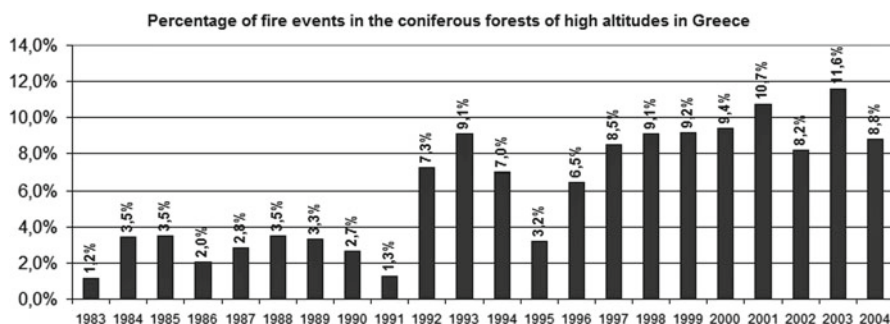
Climatic extremes observed recently (Founda and Giannakopoulos 2009; Tolika et al. 2009) have been clearly related to global warming. In particular, heat waves, when combined with droughts, can result in severe forest fires (Good et al. 2008; Moriondo et al. 2006; Rosenzweig et al. 2007). According to Dury et al. (2011) regions with more severe droughts might also be affected by an increase of wildfire frequency and intensity, which may have large impacts on vegetation density and distribution. For the Mediterranean Basin, the area burned can be expected to increase by a factor of 3–5 at the end of the twenty-first century, compared to present. Projections for 2070–2100 confirm a significant increase of fire potential for Europe, an enlargement of fire prone areas and a lengthening of fire season (Lavallo et al. 2009). Fire danger, length of the fire season, fire frequency and fire severity are very likely to increase in the Mediterranean, as effect of climatic change (Alcamo et al. 2007). Fire is a widespread process in the earth system and plays a key role in ecosystem composition and distribution (Bond et al. 2005). Fire has long ago been considered as a natural phenomenon, largely incorporated in Mediterranean climate systems evolution, having shaped their diversity (Cowling et al. 1996) and function (Rundel 1991). However, the Mediterranean Basin includes vegetation types that are either above the Thermo-Mediterranean climate zone or belong to non-fire prone ecosystems (such as sand dunes), which means that some types are not adapted to

fire and cannot cope with it, since their biota had not been subject to its recurrent action through evolution. Because of differences in post-fire survival and regeneration strategies, a change in the future fire regimes will favour some Plant Functional Types (PFTs) such as resprouters and may cause a shift in forest composition (Weber and Flannigan 1997). The rapid response of fire regimes to changes in climate can potentially overshadow the direct effects of climate change on species distribution (Dale et al. 2001); actually climatic change leads to variations in the fitness of some species too, as a consequence of significant shift of timing in their phenological phases, with advance of spring phases and extension of the growing season (Menzel et al. 2006).

### 11.3 Case Studies

Forests that are already moisture limited (Mediterranean forests) or temperature limited (boreal forests) will adapt with more difficulty to climate change than other forests (Alcamo et al. 2007). For this reason, forest types which now grow in moderately fire prone areas could face unexpected difficulties, and gradually will become fire threatened, even though they apparently are fire adapted for some traits (e.g. good sprouting ability, serotiny).

Mountainous forests receiving high rainfall amount had not been subject to recurrent fires. However, a changing pattern is documented for some regions (e.g. Greece), where data indicate that both the numbers of fires occurring at the high altitude coniferous forests and the respective area burned during the last 20 years have increased (Arianoutsou et al. 2008, Fig. 11.2). A significant proportion of these areas have been assigned a protected area status, because of their biodiversity components. Yet, the dominant tree species in high altitude areas exhibit rarely, if ever, any specific adaptation mechanism to cope with fire, as fire has not played a selective role as a disturbance factor in their evolution (see case studies to follow).



**Fig. 11.2** Number of fires occurred during the period 1983–2004 in the high altitudes coniferous forests of Greece. Note the increase observed since early 90s. (Source: adapted from Arianoutsou et al. 2008)

This situation provides both a challenge and an opportunity for the research and conservation agendas. As the biota face an increasing threat under global warming and subsequent climate change, the priority of these areas for conservation will increase.

In this chapter we will, therefore, deal with rare forest types which are expected to become more fire prone. Some of them exhibit adaptations to fire, which are however tuned to a certain fire regime different from the new, climate change-induced regime. Others do not even have any adaptation to fire. Among those, we have considered to include as study cases, forest types which are geographically restricted in the Mediterranean region, and may also consist of endemic tree species, namely *Abies cephalonica* and *Abies pinsapo*, *Juniperus macrocarpa*, *Quercus trojana*, *Tetraclinis articulata* and *Pinus leucodermis*. They are considered hotspots, in the commonly accepted sense of areas of specific high level of biodiversity, not in the sense of biodiversity hotspots at a planetary scale.

### 11.3.1 *Abies* Forests

The biodiverse mountainous areas of the Mediterranean region have been a refuge for certain conifer taxa (species of *Abies*, *Cedrus*, *Cupressus*, *Juniperus* and *Pinus*) during the glacial periods (Bennett et al. 1991). Some of these taxa have been widely used as sources of wood and food (Farjon et al. 1993) and, as a result, many of these conifers have been overexploited and are now of considerable conservation concern. Most of the conifer forests in the southern Mediterranean are threatened as a result of historical or current deforestation and overgrazing (Barbero et al. 1990). *Abies* forests represent such a type of threatened ecosystems. Eight *Abies* taxa occur in the Mediterranean Basin (Farjon and Rushforth 1989), which is one of the distribution centers for the genus (Parducci 2000). *Abies nordmanniana* subsp. *equi-trojani* and *Abies borisii-regis* require management plans to guarantee their survival (Quézel and Barbero 1990). In northern Sicily *Abies nebrodensis* has been reduced to 29 individuals (Parducci et al. 2001) and the species is listed as Critically Endangered in the IUCN Red List (Farjon et al. 2006). Of the Mediterranean fir taxa, *Abies nebrodensis* (Lojac.) Mattei and *Abies cephalonica* Loudon are the only two with island populations too. *Abies nebrodensis* is an extremely rare species with very limited distribution in the Madonie range of Sicily (Morandini et al. 1994; Parducci 2000), while insular populations of the Greek endemic *A. cephalonica* (Greek fir) occur in two islands, Euboea in the Aegean Sea and Cephalonia in the Ionian Sea. *A. numidica* in Algeria and *A. pinsapo* var. *tazaotana* in Morocco are categorized as Vulnerable (Farjon and Page 1999). Although the other Mediterranean firs are categorized as Low Risk (i.e. they have been assessed and found not to be in danger of extinction), they still face the threats common to all Mediterranean mountain conifer forests, i.e. the combination of felling (often illegal), livestock raising, farming and, more recently, devastating fires.

Several researchers point out a comparatively low fire danger in *Abies* dominated forests (e.g. Wein and Moore 1977, 1979). The main reason for this is that they mostly occur in areas with moderate to high rainfall. The high relative humidity of the air, along with the rather dense canopy of trees which intercept radiation, results in relatively high moisture contents of the forest floor. This makes fire ignition and spread less probable. However, there is some evidence that, under certain conditions, *Abies* forests are susceptible to severe fires. A prolonged drought period may be a prerequisite for such fires in fir forests. In contrast to broad-leaved and pine forests, where self-pruning is common, little branch-pruning does occur in *Abies* stands. Thus, there is continuous fuel from the forest floor to the tree crown, which increases the probability of crowning should a fire occur (Furyaev et al. 1983).

### 11.3.1.1 *Abies cephalonica* in Mt. Parnitha National Park, Greece

Among the *Abies* taxa occurring in the Mediterranean Basin, the endemic to Greece *Abies cephalonica* Loudon (Greek fir) extends in central and southern mainland Greece and in Cephalonia, an island at the Ionian Sea. On the Greek mainland, a series of intermediate *Abies* forms occur, belonging to the putative hybrid species *Abies borisii-regis*. At the northern limit, the hybrid populations mostly resemble *Abies alba* and grow together with individuals of this species, while at the southern limit they mostly resemble *A. cephalonica* (Barbéro and Quézel 1976; Mitsopoulos and Panetsos 1987; Fady et al. 1991). *A. cephalonica* forests are included in category 6.10.6 'Mediterranean and Anatolian fir forest' in European Forest Types classification scheme (EEA 2007).

In Greece, fir forests of *A. cephalonica* and *A. borisii-regis* form pure stands managed for maximizing timber production and improving its quality in conjunction with optimizing their environmental benefits. Sylvicultural measures aim at converting their current structure of even-aged stands, partly due to the irrational use applied in the past (e.g. clearings, illegal cuttings and overgrazing) to uneven-aged stands, where all tree ages are present and continuous natural regeneration occurs (Dafis 1988). In several places, however, fir forests suffer from diseases that cause high tree mortality attributed to environmental changes during the last five decades (Raftoyannis and Radoglou 2001; Tsopeles et al. 2001).

The year 2007 will be regarded as a landmark in the environmental history of modern Greece, when more than 270,000 ha were affected by fire, with most of the area (70%) being burned by only 7 fire events (mega-fires) (Camia et al. 2008a). Both ecosystems of the thermo-mediterranean climate and high altitude forest ecosystems were burned. Among the most affected forest types were those of the Greek endemic *Abies cephalonica* Loudon (Greek fir) (Arianoutsou et al. 2009, 2010). Greek fir is vulnerable to fire as it does not produce serotinous cones and does not maintain a canopy seed bank when summer wildfires occur (Habrouk et al. 1999; Politi et al. 2011). Therefore, their natural post-fire recovery is limited, and strongly dependent on seed dispersal from neighboring unburned individuals or patches (Arianoutsou et al. 2009, 2010).

Mt. Parnitha, the highest (1,413 m) and most extended mountain of Attica, in central Greece, is a National Park since 1961 and it is included in Natura 2000 “network of sites for the conservation of species and habitats” (Directive 92/43/EEC). The core zone of the National Park comprises the highest peaks of Parnitha, an area of ca. 3,800 ha, 90% of which had been covered with the endemic *Abies cephalonica* forest till 2007 (Fig. 11.3 – upper image). The altitudinal zone of the area is between 900 and 1,400 m and the climate is characterized by cool summers (usually air temperature does not exceed 18°C) and winter temperatures frequently near 0°C. The buffer zone of the Park is covered mostly by *Pinus halepensis* forest which stretches down to the foothills of the mountain (Amorgianiotis 1997). Snow is also frequent in the fir forest. One of the largest fires in the history of the Park took place in June 2007 and burned a great part (2,180 ha) of the strictly protected area and almost 50% of the *Abies cephalonica* forest. Fire did not burn the area in a homogeneous way, probably because of the dissected landscape physiography, the prevailing meteorological conditions and the tactics applied during the suppression phase. As a result, several unburned patches of various sizes have remained inside the burned area (Fig. 11.3 – lower image).

A field study was conducted during the 2nd post-fire year aiming at documenting the potential post-fire recovery in the burned fir community. No seedlings of *Abies cephalonica* were observed.

In *A. cephalonica* seeds mature during summer to early-autumn, dispersal usually begins in October, and dissemination is completed within 1 week (Politi et al. 2011). This means that a fire occurring in summer may easily consume the soft scales of the cones and the immature seeds they contain. Dispersed seeds may be buried under the organic soil or snow forming a transient soil seed bank during winter. Seedling emergence starts in late spring (Politi et al. 2009). Therefore, any seedling that might have appeared will be burned even by a surface fire.

The lack of active post-fire natural regeneration observed in the Greek fir may become worse over the long term given the masting behavior of *A. cephalonica* (Politi et al. 2009). The annual variability of cone production (masting) in populations of *A. cephalonica* has been reported following observations in northern Greece (Panetsos 1975; Dafis 1986) and it has been recently documented for *A. cephalonica* population at Mt. Ainos National Park (Cephalonia, Greece) by Politi et al. (2011). Although no *A. cephalonica* seedling was recorded in the burned sites, 2 years after fire a considerable amount of seeds was counted on the ground at distances varying up to 70 m from the adjacent unburned individuals (Fig. 11.4).

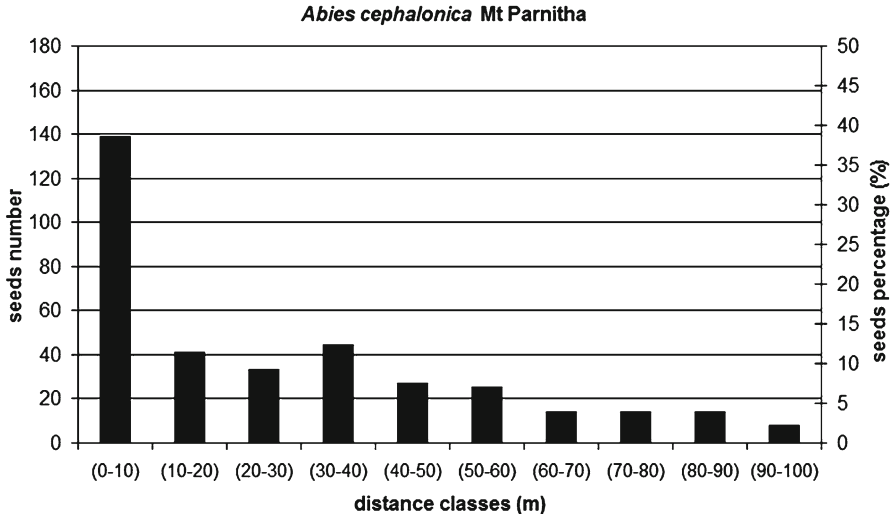
Nearly 90% of the seeds were recorded at distances  $\leq 70$  m from unburned trees, but some seeds were found up to 100 m from the unburned patch (Arianoutsou et al. 2009). The highest proportion of dispersed seeds (~90%) consisted of sound seeds. Density of seeds on the burned ground reached only 1/10 of that recorded on the floor of the unburned forest ( $1.05 \pm 0.51$  versus  $16.97 \pm 13.05$  per m<sup>2</sup>).

*Juniperus oxycedrus*, the main shrub species of the understory also failed to resprout or germinate as it has been previously reported for other ecosystem types too (Kazanis and Arianoutsou 2004; Pausas et al. 2008). However, a considerable number of species regenerated in the burned forest community. Most of the species

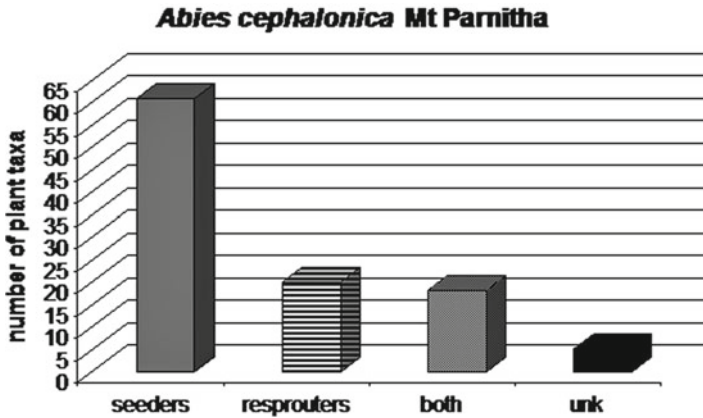




**Fig. 11.3** Forest of *Abies cephalonica* Loudon in Mt. Parnitha National Park before (*upper image*) and after fire (*lower image*). Small unburned forest patches are shown in the upper right corner of the lower picture. (Source: Margarita Arianoutsou, University of Athens)



**Fig. 11.4** Number of seeds counted on the burned ground along an increasing distance from unburned *A. cephalonica* individuals in Mt. Parnitha National Park. (Source: adapted from Christopoulou et al. 2008)



**Fig. 11.5** Regeneration mode of taxa appearing at the post-fire regenerating community of *A. cephalonica* forest in Mt. Parnitha National Park. (Source: adapted from Arianoutsou et al. 2010)

were annuals, which are absent from the understory layer of the unburned forests. The vast majority of the regenerating species recorded were seeders, many of which (32 taxa) seem to colonize burned areas through seed dispersal (Fig. 11.5). All colonizers are either annual or perennial herbs belonging to the families of Asteraceae and Poaceae.

Post-fire management was designed and performed by Park Authorities. Two main measures were applied: salvage logging and construction of wooden dams,



**Fig. 11.6** Extended log dams have been constructed following the contours after salvage logging (*upper image*). Planting of *Abies* saplings has been performed among the dams created by the logs or branches. Quite often cloth shelters have been used to protect the saplings from desiccation due to direct sunlight exposure (*lower image*). (Source: Margarita Arianoutsou, University of Athens)

and tree plantings (Fig. 11.6). Most standing burned trees were harvested and their trunks were used to construct series of log dams at a density of  $\sim 6.4$  m/ha of dams made of the tree trunk. Log dams were placed even in flat areas. In general, the efficiency of this technique decreased with the formation of gaps between their placements and the ground surface and the total distance from the point where

runoff started (Reppa, Detsis, Efthimiou, pers. comm.). Reforestation management consisted of plantings of *Abies* saplings, together with *Pinus nigra* and *Quercus pubescens* saplings. Forty five thousand (45,000) fir saplings, 8,000 Black pine and 2,000 oaks were used, while 2,000 Plane trees (*Platanus orientalis*) were planted along the streams in spring 2008. The evaluation of these plantations was made in autumn of 2009 in terms of individuals' growth and survival. Survival of the *Abies* plantings ranged between 45% and 84% and it varied as a function of aspect (Theodoropoulou, Detsis, Efthimiou, pers. comm.). Early results also suggest that when fir saplings were planted next to naturally resprouting individuals of *Arbutus andrachne* or *Quercus ilex*, survival was higher, while this effect was not observed when fir saplings were planted next to planted *Q. pubescens*. Certainly, these results are preliminary and do not cover an adequate time or space scale. More work is needed to formulate guidelines on post-fire management of these fire-vulnerable forests.

Time is certainly a critical parameter, as if the system was allowed to recover naturally, it would predictably have needed a 100 years. On the other hand, intervening in this process is not guaranteed to be a wise option. For example, extreme disturbance, like log removal, may indirectly cause soil degradation and encourage the invasion of alien species, e.g. *Robinia pseudoacacia*, which is usually planted along the roads for ornamental purposes. Constructing log dams is not necessary on flat areas and disturbance of the fragile burned soil may be another negative impact. Selection of planting material is also critical for ensuring the biological integrity of the species. If, for example, seeds used for producing the plantings are from another province, then genetic contamination becomes a potential risk.

### 11.3.1.2 *Abies pinsapo* in Spain

*Abies pinsapo* Boiss. is a conifer species of the Mediterranean Basin. It is considered a relict fir living in the south-western area of the Iberian Peninsula where it is endemic (Arista et al. 1997). According to the classification of the European Environmental Agency (2007) it is included in the category 6.10.6 'Mediterranean and Anatolian fir forest'. *Abies pinsapo* is found exclusively in the western area of the Spanish Betic Cordillera (Serrania de Ronda, Sierra Bermeja and Grazalema) frequently in shaded locations with northern exposure (Ceballos and Ruiz de la Torre 1979) in rocky places or not very deep stony soils and on limestone (European Environmental Agency 2007). It can be found over a broad elevation range, usually between 1,000 and 1,800 m, occupying 2,350 ha (Linares and Carreira 2006) of which 300 ha were included in UNESCO Biosphere Reserve in 1977. The *A. pinsapo* forests in Sierra de las Nieves were considered as relict by UNESCO Biosphere Reserve in 1995 and subsequently, they were included in Annex I of the Habitats' Directive (92/43/EEC) which protects the three main habitats in Spain. *A. pinsapo* occurs naturally in low aridity locations (Aussenac 2002) with average annual rainfall higher than 1,000 mm but nevertheless it has to cope with drought periods, as it needs more than 100 mm during summer (Ceballos and Ruiz de la Torre 1979). Usually, the Pinsapo fir forms monospecific forests although in the lower altitudinal range it can occur mixed with drought tolerant oaks (*Quercus faginea*, *Quercus ilex*



**Fig. 11.7** *Abies pinsapo* and *Quercus alpestris* mixed forests 4 years after fire. Naturally recovering vegetation of oak sprouts and companion shrubs. (Source: Jose Antonio Carreira, University of Jaen)

and *Quercus suber*) and other conifers such as *Pinus pinaster* or *Pinus halepensis* (Ceballos and Ruiz de la Torre 1979).

In the absence of fire or under very low fire recurrence, usually of low intensity, mixed uneven-aged *A. pinsapo* stands are established (Vega 1999). The low resilience of this species to fire has been confirmed in several studies and its presence has been used as a bio-indicator of non-fire-prone areas (Cabezudo et al. 1995). Several losses of *A. pinsapo* stands due to wildfires have been recorded in the last decades (Esteban et al. 2010), despite the fact that the species is not particularly flammable. Pinsapo fir shows no resprouting ability and has low regeneration success in burned and shaded areas. This has gradually induced the degradation and range contraction of Pinsapo fir forests (Rodríguez-Silva 1999), which are often replaced by more resilient species in areas with increasing fire recurrence.

The most important forest fire events in Pinsapo fir forests were recorded in the summers of 1966 and 1971 burning more than 200 ha of Pinsapo fir which has totally disappeared. In 1991, a large wildfire consumed 10,000 ha of maritime pine forest (*Pinus pinaster*) in Serrania de Ronda and burned the surrounding mixed forest including Pinsapo fir (in its lower elevation range) and also isolated Pinsapo fir woods developing in shaded and humid areas (approximately 30 ha in total) (Valladares 2009). Pinsapo fir stands showed some recovery, even if low, due to seedling recruitment in the areas of higher altitude originating from adjacent unburned Pinsapo fir stands (Arana et al. 1991).

In summer 2004, a fire burned over an old mixed stand with *Quercus alpestris* in Sierra Bermeja (Fig. 11.7). Since then no regeneration has been recorded mainly

due to the lack of specific post-fire strategy of the species coupled with the sunny exposure and the poor soil quality of the site (Esteban and De Palacios 2007).

Climate change seems to induce smaller-sized trees and leads to a greater vulnerability to pests, as observed for Pinsapo fir forest in Spain and northern Morocco (Esteban et al. 2010). The increasing temperature and changes in rainfall dynamics in southern Spain (Parry et al. 2007) induce decline and dieback of the Pinsapo fir population (Linares et al. 2009b), decreasing the area covered by the species at the lower altitudes (Linares et al. 2010; Genova-Fuster 2007). Changes in fire regime, mainly the increase in number and recurrence of fire episodes, is the major problem for these forests (Vega 1999) which have been usually under-managed (or non-managed) and disappear in areas of low elevation due to the negligible recruitment (Arista 1994). These remaining forest stands became more closed and with lower canopy structural diversity, thus increasing their vulnerability to fire. The adaptation to climate change requires a shift to proactive management, directed towards the enhancement of canopy structural diversity at both stand and landscape levels (Carreira et al. 2008). The Second Programme for the Recovery of *A. pinsapo* (Andalusia Regional Government) implemented low-intensity thinning practices at low elevation sites (1,200 m), in order to achieve the structural objectives. This improved resistance to pests and promoted natural regeneration, including the establishment of *Quercus faginea*, *Q. rotundifolia* and *Pinus halepensis*, which increased plant diversity and resilience in the treated areas (Linares et al. 2009a; Carreira et al. 2008). The resistance to drought and pests has been increased by high intensity of thinning in the more humid areas of Spain (Lietor et al. 2002).

In other areas with no post-fire regeneration, restoration was carried out with reforestation using Pinsapo fir but it showed a very low success. Esteban et al. (2010) found out that at least 2 years of high rainfall and short, mild summers after seed dissemination or planting were needed for successful natural regeneration or reforestation. Grazing control and fire prevention to promote the natural regeneration have been suggested as proper management schemes (Arista et al. 1997). To promote seedling recruitment outside the closed forests, supplementary pollination management and artificial seed rain using vigorous seeds (including seedling protection from herbivores) have also been suggested (Arista 1995; Arista et al. 1997).

### 11.3.2 *Juniperus macrocarpa* Woods in Italy

*Juniperus macrocarpa* was formerly considered as one of the four subspecies of *J. oxycedrus* (Farjon 2005), named *J. oxycedrus* subsp. *macrocarpa* (Sibth. et Sm.) Ball 1878. Based on leaf essential oils and random polymorphic DNA amplifications analysis, Adams (2000) suggested that *J. oxycedrus* subsp. *macrocarpa* merits recognition at the specific level as *J. macrocarpa*, and recovered (Adams 2011) the original name of *Juniperus macrocarpa* Sm. which was originally published in

J. Sibthorp & J.E. Smith (1816 Fl. Graec. Prodr. 2: 263) and is now widely accepted (for instance in Germplasm Resources Information Network (GRIN)).<sup>1</sup>

GRIN reports *J. macrocarpa* as native in Northern Africa (Algeria, Morocco Tunisia), Western Asia, Turkey, East Europe (Ukraine – Krym), South-Eastern Europe (Albania, Bosnia and Herzegovina, Bulgaria, Croatia, Former Yugoslavia, Greece, Italy, Malta), South-Western Europe (France, Spain), where the maritime juniper woodlands of *Juniperus macrocarpa* with *Juniperus phoenicea*, represent the mature ecosystem on outer dunes and cliffs (Adams 2011; Muñoz-Reinoso 2004).

Sand dune juniper woods are described as a priority habitat in Annex I of the Habitat Directive (92/43/EEC), under the code 2250\* Coastal dunes with *Juniperus* spp<sup>2</sup>; they are included and in several classification systems such as CORINE (16.27 – Dune juniper thickets and woods) and European Forest Types (6.10.7 Juniper forest Woods dominated by *Juniperus* spp. of the Mediterranean and Anatolian mountains).

*Juniperus macrocarpa* lives exclusively in littoral zones, preferring sand dunes, where it can form single species stands (Fig. 11.8). Traditionally, *J. macrocarpa* was used in construction for its rotting and woodworms proof wood. Some use is documented for sculpture and as Christmas tree in Basilicata (province of Matera), and Apulia (Gargano promontory, province of Foggia) regions of Italy.

Oil (for veterinary medicine, soap, perfumes, detergents and disinfectants; FAO 1995) is extracted from the heartwood of *Juniperus macrocarpa* (Valentini et al. 2003; Velasco-Negueruela et al. 2005; Massei et al. 2006). The female cones (berries) are also sued in spirit (gin) production. Juniper woodlands have a high ecological value in relation to their sand retaining ability, their ability to withstand wind effects, drought, salt sprays and increased soil pH. They serve as habitat for many plant and animal species.

During the last century, juniper woodlands were almost destroyed or profoundly disturbed by logging, urban development, pine plantations, invasions by alien species and cultivation (Muñoz-Reinoso 2003, 2004; Picchi 2008). Air pollution and increase of atmospheric particulates are also considered to negatively interfere with pollination and consequently, with the reproduction and survival of these species (Pacini and Piotto 2004; Mugnaini et al. 2004).

<sup>1</sup> USDA, ARS, National Genetic Resources Program. *Germplasm Resources Information Network - (GRIN)* [Online Database].

<sup>2</sup> Juniper formations [*Juniperus turbinata* ssp. *turbinata* (= *J. lycia*, *J. phoenicea* ssp. *lycia*), *J. macrocarpa*, *J. navicularis* (= *J. transtagana*, *J. oxycedrus* ssp. *transtagana*), *J. communis*] of Mediterranean and thermo-Atlantic coastal dune slacks and slopes (*Juniperion lyciae*). *Juniperus communis* formations of calcareous dunes. This habitat type includes the communities of *J. communis* from the calcareous dunes of Jutland and the communities of *J. phoenicea* ssp. *lycia* in Rièges woods in the Camargue.



**Fig. 11.8** *Juniperus macrocarpa* stand on dunes in Apulian coast (*upper image*) and detail of an old *J. macrocarpa* on dunes with evident signs of trampling (*lower image*). (Source: *upper image*: Luigi Forte, University of Bari. *Lower image*: Vittorio Leone, University of Basilicata)

The species exhibits, in normal conditions and in absence of fire, a low regeneration potential with a total number of juvenile plants rarely reaching 10% of the adult individuals. Seed germination could be promoted by fire smoke which has been recently reported to act as a dormancy breaking cue in some Mediterranean species



(Crosti and Piotto 2006; Crosti et al. 2006). Production of seeds is low in *J. macrocarpa* and it is accompanied by high proportion of aborted female cones (Klimko et al. 2004; Juan et al. 2003). Furthermore, the number of sound, healthy seeds is very low (Juan et al. op. cit.). Seedling survival in the field is very low, even in the absence of fire, possibly due to abiotic stress (water, wind and salinity), intraspecific competition, grazing and/or trampling (Delipetrou 2010), (Fig. 11.8 lower image). On the other hand, *J. macrocarpa* propagates readily on sand dunes by adventitious root formation but this reaction and its possible relation to sand burial, as it has been shown for other sand dune trees (Dech and Maun, 2006) has not been studied. No particular silvicultural treatment of *J. macrocarpa* is recommended, as far as we know. Due to the small size of such woods, the harvest pattern, if any, was merely an occasional, uncontrolled tree logging, pinching here and there single trees of some economic interest, when needed, presumably the biggest ones, therefore males which grow faster than females (Massei et al. 2006).

*Juniperus* spp. forests in general, but mainly *J. oxycedrus* have little resistance to fire disturbance, therefore being associated with low incidence and recurrence of fire (Fernández-González et al. 2005). *Juniperus macrocarpa* is among the species for which fire is considered a completely destructive event (Piotto and Di Noi 2001). It has no adaptive trait to cope with fire, whereas, because of its dense evergreen foliage, full of aromatic substances (terpenes) it is particularly flammable.

Post-fire regeneration of *Juniperus* species (*J. oxycedrus* in western Central Spain, *J. phoenicia*, *J. communis*, *J. thurifera*) is reported as poor by Moreno (2010). The same is likely for *J. macrocarpa* for which no data are available. Post-fire establishment for *J. macrocarpa* does, in any case, depend on seed dispersal from unburned areas and the seeds have relatively poor germination rates, never exceeding 20% (Crosti and Piotto 2006; Picchi 2008; Thanos et al. 2010).

*J. macrocarpa* woodlands exhibit a rather reduced extent, all over the Mediterranean basin, as a consequence of human pressure, mainly on coastal areas: direct human activities, in which fire is a more and more predictable threat as related to accelerated climate change, literally squeeze the relict habitat. Possible guidelines for post-fire management could therefore be based upon the plans for the recovery of *J. macrocarpa* 2,250 priority habitat (Habitat Directive 92/43/EEC). The most relevant and complete is JUNICOAST, in the island of Crete (Delipetrou 2010; Kazakis et al. 2010; Thanos et al. 2010) and the one carried out in the region of Veneto, in Italy (Fiorentin 2006), both in the mainframe of LIFE Nature actions.

Juniper seeds collected from local plants on site should be planted in deep pots so as to permit sound root growth and to facilitate the placement of saplings in the dunes. The seedlings must be at least 30 cm high before being planted, the best results to get the plants established are achieved when about 1/3 of the aerial part of the plant is buried (Picchi 2008). Planting of first-year seedlings is not recommended as they seem to have low survival rate. A more efficient solution is planting of juvenile plants (1–2 years old), originating from seedlings and/or graft offshoots taken from the respective site and grown or rooted in a nursery for several months (Thanos et al. 2010). Young junipers should be planted in autumn, after the first rains. Small young plants should be planted in the most exposed areas since they are better able

to survive (e.g. seafront of dunes), but they need to be protected with shelters. The biggest and oldest specimens (1 m/5 years) should be planted more landward, where protection from other species is greater. Wide spacing (not less than  $2.5 \times 4$ ) is recommended (Fiorentin 2006). The ratio between young and mature individuals should be greater than 1/10 while female/male individuals' ratio in the population should be kept at 1:1 (Thanos et al. 2010).

### 11.3.3 *Tetraclinis articulata* in Spain

*Tetraclinis articulata* (Vahl) Master is an evergreen coniferous tree of the Cupressaceae family known as Mediterranean sandarac-cypress, Araar or Sictus tree. The species is native from north-western Africa, in the Atlas Mountains of Morocco, Algeria and Tunisia, its distribution range covering more than  $10^6$  ha (Arman 1988). In Southern Europe small outlying natural populations can be found in Malta and near Cartagena in SE Spain, although it has been reintroduced in Seville and other provinces of the country (Mañez et al. 1997; Simón 1996). According to Baonza (2010), *T. articulata* was naturally distributed in several areas in southern Spain but was extinguished during the fifteenth and sixteenth centuries due to overgrazing, high fire recurrence and clear cutting for timber. In the first half of the twentieth century, the *T. articulata* population in south-eastern Spain was reduced to less than 15 individuals due to habitat conversion to agricultural land, the high intensity of logging for timber, overgrazing and high recurrence of forest fires (López-Hernández 2000). However, protection by proper legislation contributed to a gradual population increase, although the limited distribution in Spain seems to be under threat given the predictions foreseen under the climate change scenario (Esteve 2009).

In Spain, the Araar tree currently covers 228 ha (Esteve 2009) with 12 different populations including a total of circa 3,000 individuals (Martín-Albertos and González-Martínez 2000). It is a relict species whose forests are considered rare or residual and are included in Annex I of the Habitats' Directive (92/43/EEC). It has been catalogued as a forest type of the class 6.10. '*Coniferous forest of the Mediterranean, Anatolian and Macaronesian regions*' according to EEA classification scheme (European Environmental Agency 2007).

It is one of only a small number of conifers able to coppice (re-grow by sprouting from stumps), an adaptation to survive wildfire and moderate levels of browsing by animals (Farjon 2005). It grows in a semiarid Mediterranean climate where annual precipitation varies from 300 to 375 mm per year. In the lowest precipitation range recruitment and growth are limited and the main companion species is *Periploca angustifolia* Labill. In areas with the highest precipitation range, *T. articulata* shows higher growth and recruitment. The main companion shrubs are *Pistacia lentiscus* L., *Chamaerops humilis* L., *Olea europaea* L. subsp. *sylvestris* (Mill.) Rouy ex Hegi & Berger, *Maytenus senegalensis* (Lam.) Exell subsp. *europaea* Boiss. and several *Rhamnus* species. It shows indifferent edaphic behaviour with preference for sunny spots although the distribution and tree density depend on the competition with *Pinus halepensis* Mill (López-Hernández 2000).



**Fig. 11.9** Araar tree resprouting after the forest fire which burned 55 ha of Calblanque Natural Park (Cartagena, Spain) in summer 1992. (Source: Juan José Martínez-Sánchez Universidad Politécnica Cartagena)

In summer 1992, a wildfire burned 55 ha in the Cartagena Mountains (Calblanque Peña del Aguila y Monte de las Cenizas Regional Park) located in south-eastern Spain (Fig. 11.9). This area is an ecotone of Mediterranean and subtropical arid climate where several iberoafricanic species are found (Nicolas et al. 2004). The area has been covered with open Araar and Aleppo pine forests. Two hundred twenty two (222) Araar trees were burned and they were subsequently monitored. In spring 1993 and 1994, the area was visited to record amount of resprouting and number of seedlings.

Fire intensity, which was visually estimated based on the amount of biomass burned and the colour of the remaining ash, varied from medium to very high. Resprouting ability was significantly related to fire intensity, although 99% of the 222 burned trees managed to resprout. The individuals affected by the higher intensity showed a lower number of resprouts produced only from lignotubers. In areas burned with a medium intensity, a higher number of resprouts emerging from both lignotubers and burned branches were recorded. Two years after fire, no Araar tree

seedlings were found and the mortality rate of the regenerating sprouts was very low (López-Hernández et al. 1995). The main negative impact of wildfire was biomass loss. This effect leads to an investment of resources in growth (low productivity rate) with subsequent reduction of seed production. This in turn induces aging of the forest, which in addition to coppicing may lead to the change of forest to a more shrubby vegetation, reducing its resilience to fire and increasing the die out risk (López-Hernández 2000). In some areas, high fire recurrence has promoted *Calicotome intermedia* C. Presl. which shows a post-fire dual strategy, recovering by both resprouting and seedling (from soil seed bank) (López-Hernández et al. 1995). Furthermore, in shaded slots the Araar stands are dominated by Aleppo pines which implies a reduction of the Araar tree cover when fire recurrence is high (Nicolás et al. 2004).

To ensure the protection and improvement of the Spanish Araar tree populations, mainly after fire, the following management was proposed (Esteve 2009; Martín-Albertos and González-Martínez 2000):

- Population monitoring;
- Grazing control, mainly in regenerating stands;
- Creating a germplasm nursery and checking provenances;
- Increasing the occurrence area of the species, protecting and recovering potential habitat areas and reforesting areas with low natural regeneration using local provenances.

### 11.3.4 *Quercus trojana* Woods in Apulia, Italy

The Macedonian oak, *Quercus trojana* Webb (= *Quercus macedonica* A. DC.), is a semi-deciduous oak of eastern distribution, commonly present in Albania, Bosnia and Herzegovina, Bulgaria, Greece, Montenegro and Turkey. The western limit of its distribution area is exclusively in Italy, in the southeastern Murge hills territories, between the regions of Apulia and Basilicata, where *Quercus trojana* is considered a transadriatic endemic (Maselli 1940). It grows on “terra rossa” soils, frequently mixed with *Q. ilex*, *Q. pubescens*, *Carpinus orientalis* (Biondi et al. 2004), locally with *Ostrya carpinifolia* and *Q. frainetto*. Locally, a subspecies with big acorns (*Q. trojana* f. *macrobalana* Gavioli 1935) can occur.

*Q. trojana* (locally called *fragno*) occupies a well defined district in the sub-region of South-eastern Murge hills, with an extent of about 13,000 ha. Its presence there is made possible by the peculiar structure of the karstic environment, consisting of platform Cretaceous limestone and dolostone covered with thin layers of Pliocene-Quaternary rocks and soils. The carbonate rock is bedded, jointed, and subject to karstic phenomena (Polemio et al. 2009), which allow some water permeating the upper layers in summer (Macchia et al. 2000). Soil moisture is actually the main limiting factor for survival (Pignatti 1998).

The presence of Macedonian oak in Apulia can be interpreted as the result of a progressive withdrawal of the species during the last Ice Age, from a certainly wider range (Schirone and Spada 1995; Bozzano and Turok 2002).

*Quercus trojana* woods are included in Annex I of the Habitat Directive (92/43/EEC) as habitat type 9250<sup>3</sup> which corresponds to code 6.8.5 (Macedonian oak forest Supra- Mediterranean, and occasionally Meso-Mediterranean woods dominated by the semi-deciduous *Quercus trojana*) in the European Forest Types classification system. The distribution range is included in a trans-regional IPA (Important Plant Area) ITA 29 (Blasi et al. 2010) and in SIC (Site of Community Interest) IT 9120002 “*Murgia Dei Trulli*”. Several shrubs appear as accompanying species.<sup>4</sup>

*Quercus trojana* usually occurs in a man shaped mosaic of scattered thickets or woods of small-medium size, rarely of considerable height (>15 m), usually appreciated for fodder production in farms with animal breeding (cows, horses and locally pigs, which eat the abundant acorn production). No example of stands of *Q. trojana* with natural structure (physical and temporal distribution of trees and other plants, sensu Oliver and Larson 1996) is known (Pignatti 1998). This means that regeneration, in the majority of cases, exclusively relies on the sprouting ability of stumps, which weakens with age.

Management, oriented to domestic fuel wood production and much less to timber, the latter limited to poles and small beams, is usually represented by pure or mixed coppices or low-forests [rarely simple coppice,<sup>5</sup> usually coppice with standards or reserves; more rarely and only in excellent, albeit rare, conditions of soil fertility compound coppice (Nyland 1996)]. In all cases structures are very simplified and rather far from the pristine multilayered mixed forests covering the area in the past. From such forests, harvesting of valuable wood of *Q. trojana* for shipbuilding by the Republic of Venice is reported but not adequately documented (Bellarosa et al. 2002).

Surface fires of low-medium intensity occasionally occur in such woodlands, where understory grazing, often with a very high stocking rate, reduces fire hazard. Consequently, grazing acts as an effective fire prevention practice, by elevating the crown base height which roughly corresponds to the browse line of animals and by decreasing surface fuel load, therefore strongly reducing the risk of crown fires.

<sup>3</sup> *Quercus trojana* woods Supra-Mediterranean, and occasionally meso-Mediterranean woods dominated by the semideciduous *Quercus trojana* or its allies (*Quercetum trojanae*) Sub-type 41.782 - Apulian Trojan oak woods]

<sup>4</sup> The main accompanying species are *Arbutus unedo* L., *Asparagus acutifolius* L., *Calicotome spinosa* (L.) Link, *Cistus creticus* L. subsp. *eriocephalus* (Viv.) Greuter & Burdet, *Cistus incanus* L., *Cistus monspeliensis* L., *Cistus salvifolius* L., *Crataegus monogyna* Jacq., *Evonymus europaeus* L., *Fraxinus ornus* L., *Juniperus macrocarpa* Sm., *Paliurus spina-cristi* Mill. *Phillyrea latifolia* L., *Pirus amygdaliformis* Vill., *Pistacia lentiscus* L., *P. terebinthus* L., *Prunus spinosa* L., *Rhamnus alaternus*, *Rosa sempervirens* L., *Rubia peregrina* L., *Ruscus aculeatus* L., *Vitex agnus castus* L., (Macchia et al. 2000). Presence of *Allium subhirsutum* L.: *Asphodelus ramosus* L. subsp. *ramosus*, *Brachypodium retusum* (Pers.) P. Beauv., *Bromus erectus* Huds., *Galium aparine* L., *Geranium purpureum* Vill., *Geranium dissectum* L.; *Teucrium chamaedrys* L.; marks anthropic influence. (Misano and Di Pietro 2007).

<sup>5</sup> Coppicing is an ancient form of woodland management, mainly directed to fuelwood production, which involves repetitive felling on the same stump, near to ground level, and allowing the shoots arising from adventitious bud at the base of a woody plant to regrow from that main stump (coppice stool).



**Fig. 11.10** *Quercus trojana* stand a few months after coppicing (*coppice with reserves*): stools in rapid growth form the sparse understory under the reserves, in number of about  $150 \text{ ha}^{-1}$  (*upper image*) the first spring after coppicing. Abundant grass understory may represent a fire hazard in this early phase of growth with low fire resistance (*lower image*). (Source: Vittorio Leone, University of Basilicata)

Situations of isolated wood fragments in a rural matrix, with high edge effects, increase fire risk.

Regeneration strategy of *Q. trojana* woods is mainly based on the high sprouting capacity, which is also the basis of coppicing. Epicormic sprouting is also present after fire (Fig. 11.10).

Seed germination is rather rare, being strongly dependent on climatic conditions and on abundant seed availability. Management of *Q. trojana* as coppice is a factor of progressive reduction of cover density, due to ageing of stumps and progressive

loss of their sprouting capacity. Improper management and excessive grazing pressure can induce different stages of degradation of coppices too, which start as high forest and could in many cases tend toward a sparse stand of old trees ending up as barren carstic rock terrain.

Post-fire treatment is normally based on local Forestry Regulations which obliges owners of broadleaved forests to cut damaged stands as soon as possible, possibly in the first autumn after the fire event, for preventing dangerous fuel accumulation, pests and noxious insects' diffusion, but above all in the search of aesthetics. This treatment, in the case of coppice, corresponds to an ahead of time harvest.

Post-fire response, based on vigorous sprouting, is crucially dependent on grazing prohibition for the first 4–5 post-fire years and in the time interval between consecutive fires; a new fire in less than 5–6 years after the previous one means total destruction of such woodlands.

A possible improvement of post-fire treatment should consider the resilience and resistance to fire of *Q. trojana* at the level of single trees, avoiding current salvation felling, but applying a very selective salvage logging, harvesting surely compromised ones and avoiding elimination of individuals with good epicormic shoots from branches, which could survive. This treatment partially produces a shift from *coppice with reserves* (from 60 to 150 plants ha<sup>-1</sup>) to a *compound coppice* in which some old standards are harvested and the remaining are left to grow for additional rotations in number decreasing with age. A longer rotation time should be also applied, at least 20–22 years.

A change of climate from wetter phases to drier ones in Southern Italy [warming in all seasons, marked precipitation decrease, long periods of drought and decrease in soil moisture (Castellari and Artale 2010)] could strongly interfere with the ecology of *Q. trojana*, representing a threat for its survival. Since a very significant increasing trend in fire danger is projected in Apulia (Camia et al. 2008b), an increment in forest fires is also expected in the distribution range of *Q. trojana*.

Increasing drought could amplify oak decline too. The oak stands of *Quercus pubescens* and *Q. trojana* in the region of Apulia show a dramatic dieback (Sicoli et al. 1998).

Such changes could cause increased *Q. trojana* mortality, reduce the occurrence of optimal conditions for germination, which are strongly related to moisture, and therefore progressively hamper species' presence. On the other hand, repeated fires and management as coppice could gradually reduce the vitality of coppices, depleting their reserves and inducing a progressive regression in the already restricted distribution range and abundance of such an interesting endemic.

### ***11.3.5 Pinus leucodermis Woods in the Pollino National Park, Italy***

Bosnian pine or palebark or white-bark pine (*Pinus heldreichii* Christ. (1863) syn. *Pinus leucodermis* Ant. (1864), or *Pinus leucodermis* (Antoine) Markgraf ex Fitschen (Richardson and Rundel 1998) is widespread in the western Balkan

peninsula (Albania and Greece) at 1,000–2,500 m elevation, in the central western part of the Balkan peninsula (including Bosnia, Serbia, Montenegro and Bulgaria), and in Italy.

In the southern Italian Apennines *P. leucodermis*<sup>6</sup> occupies dry, sunny sites from 530 up to 2,240 m elevation, forming mixed stands with silver fir (*Abies alba* Mill.) and European beech (*Fagus sylvatica* L.) or pure stands on steep and dry rocky southern slopes (Avolio 1984, 1996). The already reduced range of *P. leucodermis* is split into five mountainous areas of Monte Pollino in the Pollino National Park (Gargano and Bernardo 2006). The scattered and isolated populations in Southern Italy of this Balkano-Italian restricted endemic are at the western limit of its geographical distribution range and represent a biogeographical island, interpreted as a relict of Tertiary flora of oro-Mediterranean forests which are genetically isolated from Greek populations (Bucci et al. 1997).

Populations are found at diverse ecological conditions, from the lower vegetation belt, where they are mixed with evergreen sclerophyllous vegetation, up to the alpine vegetation belt beyond the closed formations of *Fagus sylvatica* (Guerrieri et al. 2008). Stands at higher elevation are under the form of open woodland, characterized by large white snags and stumps mixed with scattered old monumental trees: such stands exhibit very low density (20–40 trees ha<sup>-1</sup>) but relevant size (dbh up to 1.20 m) and age until 900 years and more (Guerrieri et al. op.cit., Corbetta and Pirone 1996). The soil is shallow, rendzina – like, with large outcropping rocks on bedrock of fissured greyish limestone (dolomite) from the Mesozoic.

Mean annual precipitation is 1,570 mm; mean annual temperature is 4°C; snow cover lasts from November to the end of May (Avolio 1996). According to the classification of Rivas-Martínez (2004) the climate varies from supra-Mediterranean up to subalpine/alpine (Gargano and Bernardo 2006).

*Pinus leucodermis* woods are included in Annex I of the Habitats Directive (92/43/EEC) as habitat type 95A0 which corresponds to 42.7<sup>7</sup> in the Palaearctic classification system and 6.10.5 (Alti Mediterranean pine forest) in the European Forest Types system. The distribution range in Basilicata is included in trans-regional IPA 30 “Pollino” (Important Plant Area) and in SICs IT 9210165, IT 9210245, IT 9210185.

White-bark pine’s populations occur in different environmental conditions (Fascetti 2001) in dry and rocky mountain grasslands on the cliffs (Corbetta and Pirone 1981) and in screes (Corbetta and Pirone 1981), (Fig. 11.11 upper image). The main accompanying species are: *Daphne oleoides*, *Rhamnus pumilus*, *Sorbus graeca*, very short shrubs of *Juniperus hemisphaerica* (20–60% in cover) and *Gentiana lutea*, surrounded by patches of grassland of the *Caricetum*–*Seslerietum nitidae* association (Bonin 1978).

<sup>6</sup> We keep the name *Pinus leucodermis* as suggested in Conti et al. (2005).

<sup>7</sup> High oro-Mediterranean pine forests Subtype 42.71 White-barked pine forests Local tree line formations of *Pinus heldreichii* restricted to the southern Balkans, northern Greece and southern Italy, usually open and with an undergrowth formed by stripped grasslands on dry, often stony or rocky soils).





**Fig. 11.11** *Pinus leucodermis* typical community (old growths, matures, over matures, hard snags prevail on scarce saplings) (*upper image*) and sapling in a safe site between rocks with low nutrient availability near tree stumps or fallen logs (*lower image*) in the Pollino National Park, Italy. (Source: *upper image*: Pietro Civale, CODRA. *Lower image*: Antonio Romano, University of Basilicata)

The main value of Bosnian pine is currently as an ornamental and landscape tree rather than as a timber species. In the past it was very important in the local people's cultures (Fascetti 2001) and the wood was used for many handicrafts (timber, windows and doors frames, furniture, boxes and frames of musical instruments, oars, boats, wood chests etc.; resin also was tapped). Relevance in local culture is confirmed by its choice as symbol of the National Park of Pollino.

Since the 16th century *P. leucodermis* forests (approximately 3,000 ha in size) were used for grazing by goats, sheep, cows and mares arrived by transhumance

from the Ionian Sea coast. Timber from *P. leucodermis* and *Fagus sylvatica* was the main resource for heating, lighting with torches, housing and dairying (Todaro et al. 2007). In the 1920s law enforcement favoured the increment of the resident population and their agricultural and pastoral activities. The number of browsing animals in the area increased till the beginning of 1970s. With the establishment of Pollino National Park in 1993, *P. leucodermis* became a protected species (Todaro et al. 2007) and grazing activities were somehow controlled.

Overgrazing has seriously affected population structure of the species with effects that are still evident. Only adult trees occur in the area and a marked lack of intermediate age classes is observed. A limited recruitment of new individuals at the tree-line ecotone is observed in many stands as a consequence of such long-lasting pressure (Morandini 1950; Longhi 1956). Bosnian pine is characterized as Low Risk in the IUCN Red List of threatened species. However, its natural distribution is currently rather limited and fragmented, mostly because of human pressure and forest fires (Vendramin et al. 2008).

Recent observations report that *P. leucodermis* forests are affected by recurrent fires in its lower elevation range but rather satisfactory post-fire regeneration occurs (Pierangeli pers. comm., Saracino pers. comm.) in more wet and favourable sites. Regeneration is ensured by seeds from adjacent areas or from surviving trees in the burned area; small or medium size trees generally do not survive fire, but old growth trees seem rather resistant to fires which are usually low-intensity ones. These observations confirm Gargano and Bernardo (2006) who uphold that in the SW part of the range and at low and intermediate elevation (900–1,300 m), fire appears to be a determining factor in establishing and maintaining young *P. leucodermis* populations (Fig. 11.11 lower image).

*P. leucodermis* exhibits great adaptability and great colonizing potential as a pioneer species. Its high level of self-fertilization, (probably due to the presence of a reduced number of recessive embryonic lethals (Morgante et al. 1994; Vendramin et al. 2008) as a consequence of genetic drift) represents a selective advantage for this highly competitive, pioneering and precocious species, (Gargano and Bernardo 2006).

To our knowledge, no specific protocol for post-fire treatment has been implemented for *P. leucodermis*. The general recommendations for post-fire management of pine forests could be also applied for *P. leucodermis* (see relevant chapters). An additional recommendation is to respect the natural patchiness of regeneration and the presence, if any, of “tree collective structure” or collectives, i.e. of a clump or cluster of seedlings which is functionally structured so that the outer trees act as protective of the inner, most fit ones. At favourable sites cluster planting can be used too, in order to minimise seedling mortality and is therefore a valuable strategy for afforestation in non-suitable sites at higher elevations (Schönenberger 2001; Souček and Špulák 2011), minimizing browsing damages. In high mountainous areas, tree collectives should be considered as resistant and stable stand elements. Because of the large gaps between the tree collectives, they radiate from very few individuals, gradually expanding around the edges by natural seeding (or layering for some species) with the result that uneven-aged cohorts are formed (Fillbrandt 1999). In the tree collectives humus development is faster. This is especially important in these habitats where the upper soil layers are subject to erosion (Ucler et al. 2007). In the

higher elevations of Pollino National Park, where the number of trees is much reduced, surface fire frequency is expected to increase with climatic change. Fire could cause rapid and drastic reduction of the population size by killing surviving adult trees with consequent further losses in seedling genetic diversity and adaptive potential (Vendramin et al. 2008). In addition, the number of already scarce sapling could be further reduced. Seedling establishment can be compromised by the high post-fire competitive pressure from herbs. In such areas, if natural regeneration is absent, a possible management option could be seeding in the microsites where regeneration generally occurs (mainly bare lithosoils strips between rock outcrops). Given the high inbreeding and low outcrossing rates of this species, ideally seeds should be collected from a large number of trees (approx. 100) and from trees growing at least 100 m apart as suggested in Euforgen guidelines (Vendramin et al. 2008). Seeding is costly but in the strictly protected zone of the National Park (integral reserve) it seems the most natural treatment possible, since no alteration of habitat is induced.

## 11.4 Key Messages

- Significant climate changes are projected by the XXI century at a world scale, with temperature increase and decreased precipitation. Such changes could affect the frequency and severity of conditions suitable for the ignition and spread of fires;
- Any change in fire regime (frequency, severity, and seasonal timing of fire) may change the ecological consequences of such fires, leading to shifts in community structure and species composition;
- Species growing in the Mediterranean region exhibit adaptations and resistance to fire that lead to predictable responses to fire, in the current fire regime, but can prove unfit to cope with significant variations. As a consequence, species and communities can be vulnerable showing decreased resilience, along with possible shifts in their distribution and performance;
- Many forest species which share this vulnerability status are high altitude species, or narrow endemics with a restricted distribution range. These species may be unable to cope with increased fire frequency, or with the simple fire occurrence, in the case of forest types in which species have no adaptations to fire;
- We discussed the management of increasingly fire-prone forest with species of high conservation value (*Abies pinsapo*), restricted or very restricted endemics (*Abies cephalonica*, *Pinus leucodermis*, *Tetraclinis articulata*), species of narrow distribution range (*Quercus trojana*) or under the threat of range reduction and anthropic pressure (*Juniperus macrocarpa*). Some of these may lack in production value, but they all considered as emblematic of high biodiversity conditions;
- For their conservation, further scientific research and the development of specific protocols for post-fire restoration are needed to cope with the threat of increased fire hazard.

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