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CLIMATE CHANGE, WILDLAND FIRES AND BIODIVERSITY IN EUROPE

Documentr prepared by
Mr José M. Moreno, University of Castilla-La Mancha, Center for Fire Research [CIFU] (Toledo)

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EXECUTIVE SUMMARY

Every year, more than 0.5×10^6 ha are swept by fire in Europe, mainly in the countries of Southern Europe and the Mediterranean (SEM) threatening human and natural values. This part of Europe hosts a large fraction of European biodiversity, being the Mediterranean one of the hotspots of biodiversity in the world.

Fire is a main factor in shaping vegetation throughout the world. Its role in determining biodiversity is probably large, although it has not been quantified. Mediterranean ecosystems have evolved in a world with fire. There are numerous plant traits that can be associated to a long history of fire. This has been confirmed by phylogenetic analysis.

Fire regimes until the middle of the Holocene were characterized by long fire cycles. This cycle was accelerated as man occupied the land. Fire became part of the management tools he used to conquer and use the territory, in addition to grazing, ploughing, coppicing, planting and other agriculture-related operations.

However, during the second half of the XX century, with economic development people fled to the cities and unproductive land was abandoned. Landscapes started to be covered with natural vegetation or plantations, often with highly flammable conifers. Fires started to become more frequent and widespread throughout the territory and in all countries of SEM. Fires are today a significant factor of extant landscapes.

Most fires are caused by people. Fires are extreme phenomena, which means that few, infrequent severe weather and climate situations can lead to major fire disasters. That is so despite the huge efforts made to increase fire-fighting capacity. Multiple very large fire episodes during which fire-fighting services are overwhelmed have occurred recently, affecting large portions of some countries or whole regions.

Fires do not burn the landscape at random, and tend to affect certain vegetation types more often than others, and occur at certain locations. Fires burn through natural protected areas as well. During the last three years, of all the area burned in the largest EU Mediterranean countries nearly 1/3 was part of the Natura 2000 network. Areas close to or at intermediate distance to roads or towns are the ones that burn most frequently. These elements of fire risk are important for conservation areas.

Although many SEM ecosystems can be considered to have evolved under fire, the current fire regime is different from what it might have been in the past. Changes in fire regime, such as increased frequency and severity of fires, threatens ecosystem stability and, in some areas, favours degradation loops that impedes the recovery of the vegetation towards more mature stages.

Postfire regeneration usually follows the autosuccessional pattern. Plants are able to withstand fires mainly by surviving the blaze and resprouting or by germinating from seeds that survive the fire as well and, in many instances, require heat-related stimuli to germinate. In a few years after fire the plant community resembles that before the burn. However, direct regeneration is not always warranted. Furthermore, there are many emblematic species that do not regenerate well after fire.

Resilience of plant species is closely related to fire regime characteristics. Changes in them can have important consequences for species persistence. For instance, too short an interval between fires will threaten the persistence of plants whose age to sexual maturity is long. When these occur in tree woodlands, as is often the case, these areas will turn into shrublands, hence changing the system for a long period of time.

Event-dependent effects are also important for plant resilience. High fire intensity and severity can impair regeneration. Similarly, large fires can reduce the colonization from the edges. Fires at different times of the year can have different effects due to changes in the physiological status of the plants or on seed availability.

Postfire climatology can also be critical for plants regenerating from seeds. This is most relevant when multiple large fires occur, since large extension will then be regenerating in parallel and subject to climate variability that operates at large scales. Changes in rainfall patterns or, particularly, droughts can affect such regenerating stands and have long-lasting effects.

Animals exposed to fire will be killed. Animals can survive the blaze by fleeing away from the fire front or seeking shelter. Animal succession varies among groups but, in general, follows the pattern of regeneration of the vegetation. Time to recover the initial community will vary depending on the type of organism, some recover quickly, others require decades.

Animals are equally sensitive to fire-regime characteristics. Short fire return intervals can affect some populations and species and delay their recovery for years. Fire size can affect animals, particularly when these must colonize from the edges or from unburned islands whose abundance might be reduced in large fires if these are more severe, as is commonly the case. Fire season can also be critical since it can affect populations at different stages in the reproductive cycle.

Studies addressing the responses of organisms to fire studies are usually done separately for different groups. Yet, a number of interactions occur and these can be important for the persistence of populations and species. Competition among plants can affect the regeneration after fire and, in some instance, lead to modifications in the type of stand. Among plant-animal interactions, seed predation is important. For some plants, missing the window of opportunity for establishment after fire can result in long term effects due to the higher percentage of seed predation. Herbivory is equally important, notably when this involves large scale forest die back that modifies fuel content and can affect fire risk and characteristics.

Other interactions such as pollination services or postfire dispersal can affect as well some regenerating populations, although much more needs to be known. Interactions among animals via predation can also be impaired by fire.

Plant and animal population and communities can be affected by postfire management. In particular, salvage logging, which is commonly practiced, can be important. Different organisms will react differently depending on their habitats preferences. Knowing which are sensitive to logging is relevant for determining postfire management.

Interactions between organisms mediated by the changes in the landscape are also important. Fires can modify the various functional groups of plants at large scales, since the sensitivity to fire varies depending on regenerating strategies, life form or distribution range. Animals may colonize areas depending on local patterns of abundance. Patterns of fire regime can modify species abundances. In some instances, increased abundance and richness of some groups (birds) has been associated to fire. So the role of fire may not necessarily be negative for all organisms.

Future land-use and land-cover is very likely to continue as a result of climate change and additional socioeconomic changes. Hence, more land will be added to that already existing in a state of abandonment, with the corresponding consequences for species, landscapes and fire.

Future climate scenarios for SEM project increases in temperature that are higher than the global mean. Rainfall is also projected to decrease and shift temporally. Additionally, and increase in the frequency and intensity of droughts and heat waves is foreseen.

Climate change poses significant threats to the persistence of plants. Niche modelling exercises show that most plant species will lose climate space, making a sizable number of them to become vulnerable or endangered by the end of the century and much irrespective of emission scenarios. Climate change poses similar threats to animals. Mammals, birds, amphibian and reptiles are projected to reduce their ranges as a result of climate change. Even though there are indications that they are responding already to the observed climate change, they are lagging behind in their response. This means that many species are already living under climate stress.

The risk of fire is projected to significantly increase in much of Europe, particular in SEM. Increases in part of the boreal are also anticipated although they are of reduced magnitude. The duration of the fire season in SEM is projected to expand, as it is also projected that the number of days of risk, particularly of very high risk, will increase. These changes are very likely to augment fire risk in most areas. The increment of weather and climate extremes is most likely to affect the probability of large fires. Interactions among increases in fire risk with other stressors, particularly drought, are most relevant for the regeneration of the system. Heat waves can add to this, and be important for animals that will have reduced shelter in open, regenerating stands.

Managing Mediterranean areas that are the result of human transformation for thousands of years is a challenge. Much more so under the evolving conditions of climate change and other global change drivers. Maintaining biodiversity, natural values and services, including their cultural landscapes, is a daunting task.

Following are some recommendations can help biodiversity conservation in a context of changes in climate, landscape and fire risk:

1. The role of fire must be included in conservation of species and habitats in fire prone areas

Fires have been occurring, and will most certainly occur within many protected areas in southern Europe and in the landscape matrix that surrounds them. Fires are generally considered as a threat, and fire suppression is the dominant policy throughout SEM. There are enormous skills and capacities to fight fires. Yet, when they break out inside or around protected areas they will burn through them. But since the main/only policy is to fight them, provisions to understand how they directly or indirectly affect protected areas and species once burned are, for the most part, lacking. Until now, the ecological role of fire is ignored. Consequently, when they occur there is no contingency plan as to how the affected system will be impacted. Therefore, even without any climate change, biodiversity conservation plans need to consider how fires will affect species and habitats throughout the territory. Fire ecology is a must in all management and conservation plans, and strategies to incorporate this knowledge must be enacted.

2 Fire can play a role for fire in conservation

Some ecosystems and species depend on fire or can benefit from it. Identifying them might be critical since current policies will jeopardize their persistence. In these cases, plans for introducing fire, either by prescribed burning, or, when appropriate, with wild fires within acceptable conditions to avoid other risks must be made. Because the prevailing view is that fires are undesired, and the risks that entail managing fires is great, conservation plans in need of fire must be implemented with great care to avoid accidents that would stop the continuation of needed plans with the concourse of fire.

3. Conservation plans aimed at specific target species must consider how fire will affect them

Species or groups of species are impacted by fire differently, depending of fire characteristics and other factors. In the case of protected areas whose objective is one or a group of particular species, the viability of their conservation in a context of fire needs to be specifically considered. Management plans that address the possible impacts of fire need to be species or group specific, since different species are likely to respond differently to fire.

4. Vulnerability of the protected areas network to fire

Corridors and stepping stones are important elements for insuring population persistence and species migration, more so in view of the impending threats. These elements, however, may be subject to fire. When these components are formed by forest, fire can alter their functioning capacity for long. Since it is very likely that some of these more isolated elements are in areas with greater human influence, their susceptibility to fire and repeated fire might be rather great and needs to be quantified since its long-term persistence may be severely threatened. As with the rest of the protected areas, the impact of fire needs to be known in advance in order to better evaluate their capacity to continue playing their role. Robust network designs, capable of not succumbing to a single fire, are needed to allow these places continue playing their vital service.

5. Urban developments and roads near protected areas threatened their persistence

Most fires are lit by people. Towns and roads are the main sources of ignitions. However, the probability of burning is still high at some intermediate distance to roads and towns since fire can travel long distances. Protected areas within these domains are at higher risk of fire than those further away. Urban developments into the wildlands and near protected areas can be a threat to these due to increase ignition probability and subsequent fire. Also, the network of roads crossing protected areas, in addition to other perils, can clearly add risk. These two elements must be cautiously considered when declaring protected spaces and be particularly monitored during the time of high fire risk. Eventually, specific restrictions might have to be put in place to minimize risks. Risk mapping of protected spaces taking into consideration proximity to roads and towns is critically needed.

6. Identifying synergies/conflicts between fire and conservation

Fire fighting includes, among other, fire break lines or fire-break areas. These can provide open space and hence favour species persistence different to those in the preserved matrix, particularly when these are forest. The role of such areas and corridors as sources of rapid colonization after fire needs to be appraised. These areas can serve as colonization points but there are positive or negative elements (increasing potential for invasive species) that need to be fully considered. The advantages and disadvantages of these areas in the event of fire need to be taken into consideration.

7. Changes in the landscape matrix through fire

Abandonment will continue in response to changes in socioeconomics and with climate change. Abandonment modifies the landscape matrix towards homogenization and that can threatened the persistence of many species. Fires can open up space and introduce large changes in the landscape matrix. Not all organisms will be equally affected but such changes in the landscape structure. Some, through the openings made by fire, will be favoured. Others will be negatively affected. Conservation plans must therefore contemplate the landscape scale changes that are introduced by fire.

8. Assessing future risks

Changes in fire frequency, intensity/severity, size and season must be specifically contemplated for conservation areas under scenarios of climate and land-use/land cover change. This must be done for current areas with fires and for those in which fires were not present but that are likely to occur due to the changes in climatic conditions and other factors. Each of the parameters that define the fire regime can differentially affect the various species. Changes in fire season, particularly when migrant species are concerned, need to be cautiously considered. Consequently, the impact of each of them needs to be assessed in general or for the particular species or group of species that are of interest.

9. Fire, species conservations and other stresses

Conservation scenarios that include fire must take into consideration the level of stress being endured by the various species since, little by little, they will inhabit areas that are more stressful for them due to changes in climate among other stressors. The capacity of particular species or groups to respond to fire under such circumstances and to changes in fire regime needs to be appraised. As fires might occur under extreme conditions not seen until now (drought being the most relevant) this type of interactions need to be fully taken into consideration in future management plans for biodiversity conservation. Additional stresses due to more frequent and intense heat waves, particularly in the open habitats of the first years of regeneration after fire, must also be known.

10. Conservation plans must include worst case scenarios

Although the great majority of fires are of small size, some of then can attain very large sizes, in the order of thousands of hectares. In Spain, the maximum size of any fire recorded is around 30.000 ha, and the maximum length is 45 km (Moreno et al. 1998). The potential for one fire to spread over a whole protected area at once is not negligible. Smaller and homogeneous areas in a matrix of high fire risk are the most threatened. The prospect of increasing fire size under future conditions further adds to this. Consequently, worst case scenarios that include burning a large portion or even the whole protected area when these do not exceed several thousands of hectares needs to be contemplated. The role of buffer zones in this context needs to be equally appraised.

11. Fires as opportunities to accommodate species to the new climate

Fires, by opening new space, and by having reduced competition among organisms in the early phases can open new space for species to move upwards or northwards in search of suitable climate. But this can also be used for invaders. Differentiating the new colonizers that are now attuned to the new conditions from those invading is important. Identifying the potential for fire to act as stepping stones must also be considered.

12. Identifying species at greater risk

Species of late successional stages, thus requiring longer time to colonize burned areas, are probably the ones at greater risk in scenarios of increased fire frequency. Moister sites should regenerate quicker than more xeric sites, but their rate of recovery will be delayed with the onset of reduced precipitations under future climate for large parts of SEM. Consequently, their recovery period will be extended and the probability of burning again in earlier stages of regeneration indicates that species proper of mature successional stages might suffer. Studies should emphasize determining which groups of species enter at which state of the postfire succession and on the time needed for their recovery.

13. Bottle-necks

Among those species most likely to suffer from fire are those of reduced distribution that are linked to particular systems that are fire sensitive. That is, those that do not regenerate after fire. A fire, particularly a large one, can severe these populations for long, making its recovery difficult. Identification of bottle-necks and deadly-traps among organisms and their systems in the event of fire is critical for those species that may be most threatened.

14 Species linked to others

Climate change is producing mismatches among species (in pollination, in dispersal). Furthermore, fire con contribute to alter them. Identifying mismatches that are enhanced by the combination of fire and climate change might be of relevance for the maintenance of species that may already be in danger.

15. Changes in the protected network

With climate change, the size of the protected areas will have to be increased to achieve the same conservation objectives. Until now, fire has not been taken into consideration in the design of the network of protected areas. Yet, its effectiveness can vary. Consequently, future modifications must consider how fire would affect its effectiveness. Since it is likely that the protected areas of the current network are those in a better state of conservation, which, presumably, are those further away from human influence, it is likely that new additions will be closer to human habitations, thence with higher risk of fire. Risk of fire must be included at the time of modifying the network of protected areas.

16. Educating on the ecological role of fire

Fire is commonly seen as something negative, but it can play a dual role in the conservation of biodiversity. Fire, for the most part, hardly receives any attention in education, even in university programs, or not as much as its relevance demands. Every effort must be done to form and inform the general public and students at all levels about the role of fire in ecosystems and biodiversity conservation.

17. Research needs

Knowledge on how fire affects the various groups of organisms across gradients is still a must. Long term observation sites, where the main groups of are studied jointly should be established.

Large fires, particularly large fires episodes, are laboratories that should be explored in depth for their role on biodiversity. Since many of these fires occur along gradients, these are opportunities that should not go by unexplored.

Maps with fire history are now possible for the last decades. These offer opportunities to study the impact of repeated fires on biodiversity across groups and across landscapes.

Protected areas are not static and will change with climate change. Modelling their fate and their vulnerability under scenarios of climate and fire change are crucial to understanding their future role in biodiversity conservation.

1. INTRODUCTION

Every year, more than 50.000 fires are recorded in Europe, burning more than 0.5 MHa (UNECE 2002, Dunnov et al. 2005, Anonymous 2006, San-Miguel-Ayanz et al. 2009). Fires occur almost in every country but are particularly important in southern Europe, from Portugal to Spain, France, Italy, the Balkans, and Greece and, to a lesser extent, Turkey. Fires are also important in some countries in central and Eastern Europe, notably in Poland, as well as in the boreal countries. Fires are funneled by the right combination of climate and weather, fuels and ignition sources. Most fires are caused by people (San-Miguel-Ayanz et al. 2009). Consequently, provided that ignitions are usually not limiting, the first two factors are critical for fire occurrence. Fires, particularly large fires, which are the most devastating ones, occur mainly in the summer. Nevertheless, in some countries fires related to pastures and management can occur in spring and fall. Actually, in some regions of Europe, fires occur during the winter, when foehn type situations propitiate their occurrence (Telesca et al. 2010).

By forest fires we usually understand any fire that occur in high forest, wooded areas, or on any wildland, irrespective of the woody cover it might have. Therefore, by forest fires it must be understood virtually any fire that propagates out of control and affect an area that is not urban or devoted to agriculture. Consequently, almost any natural area that carries vegetation that at some point in time has low moisture content, can be affected by fires provided the right weather and climate conditions and an ignition source.

Fires consume dead and living plant materials, litter and organic horizons, and, except in the case of animals that flee away rapid enough from the propagating front, or seek shelter underground or elsewhere, all groups of animals. Fires also affect microorganisms of the upper soil horizons either directly, through the heat, or indirectly by the change in conditions and resources they bring about in soils (Whelan 1998). Consequently, fires can affect all groups of land biodiversity. Fires also affect riparian vegetation and through this and other indirect effect on the quality of water can also affect the freshwater biota (Spencer et al. 2003). Since many of the organisms populating these habitats are under various status of conservation, understanding the threats to biodiversity requires that we comprehend the role of fire in the systems they inhabit.

But the effects of fire go beyond the specific effects on a given ecosystem. Fires also change landscapes by creating a mosaic of patches of different sizes and forms and time elapsed since fire. By modifying these characteristics they influence habitat configurations and through that dispersal and colonization patterns as well as other ecosystem processes (Turner 1989).

Fires are occurring now almost in any country in Europe. However, the mild and wet winters, warm and dry summers, moderately fertile soils and moderate to high primary productivity in much of southern Europe, notably in Mediterranean-type climate countries, provides an almost perfect combination for fires to occur during the summer. The variety of climates, substrates, soils and land forms of the Mediterranean Region (MR) has been molded by man since millennia by using, among other management tools, fire to produce highly diverse landscapes (Naveh 1994). In addition, biodiversity in this region is among the highest in the world, and second only to the tropics (Myers et al. 2000). Furthermore, climate change is projected to significantly impact this region, by increasing temperatures above the mean at greater rates than the global average, decreasing precipitation and increasing climate-related extremes, among other (Alcamo et al. 2007). All of these factors will directly and indirectly affect biodiversity in the region. Consequently, southern Europe and the Mediterranean (SEM) will be the main focus of this report.

2. WHAT DO WE KNOW ABOUT FIRES FROM THE PAST?

2.1 Fire and biodiversity

Fire is considered to be a main factor in shaping extant vegetation in the world (Bond and Keeley 2005). Many areas in the world host vegetation and landscapes whose potential to sustain biomass is much greater than what it is realized. The implications of the mismatch between actual biomass and potential biomass based on extant climate has multiple implications since such changes in vegetation structure can hardly be conceived without taking into consideration their cascading effects on species composition (Bond and Keeley 2005). The consequences of fire on biodiversity can better be understood by looking at the effects on species composition when fire suppression is enacted on ecosystem with a long fire history. Losses of as much as 60% of the plant species have been reported as a consequence of fire suppression in remnants prairies (Leach and Givnish 1996). Similarly, in fire-driven ecosystems changes in fauna are expected in the absence of fire (Lyon et al. 2000). The implications are much larger when the presence or absence of fire implies a whole change in biome, as is the case of many African savannas (Sankaran et al. 2005). Fire, therefore, is a most powerful agent of plants and animals change and its role on controlling extant biodiversity is paramount, although it has yet been poorly quantified.

2.2 Fire footprints

Fires have been present on Earth since the atmosphere accumulated enough oxygen to sustain combustion (Chaloner 1989). Lightning is the main source of natural ignitions (Price and Rind 1994). The possible role of fire as an agent of evolutionary change can be inferred from the presence of traits that increase plant persistence in fire environments. These include the very thick bark of some tree species, like *Quercus suber* L. (Zedler 1995), or *Pinus* (Tapias et al. 2001), the serotiny of many *Pinus* species (Thanos and Daskalakou 2000, Tapias et al. 2001), the presence of permanent soil-seed banks that are activated by heat (Trabaud and Oustric 1989, Luna et al. 2007); or other fire related cues, including smoke and the chemicals it contains (Moreira et al. 2010, Nelson et al. 2010), or the development of a canopy architecture that favors flammability (Bond and Midgley 1995, Schwilk 2003), among other. Phylogenetic analysis of plant species confirms that the evolution of plant regeneration traits (resprotuing, seeding) in the Mediterranean appears to have been driven by fire (Pausas and Verdu 2005). Furthermore, plant communities in the MR appear to have been sorted out by fire through a filtering process whereby species with fire-persistent traits are more abundant (Verdu et al. 2007, Pausas and Verdu 2008). The filtering process is strong enough as to act even at local scales in relation to, for example, soil characteristics (Ojeda et al. 2010).

2.3 Palaeohistory of fires

Fires are thought to have naturally occurred in the Mediterranean since at least the Miocene (Dubar et al. 1995). The earliest convincing evidence for human control of fire dates back 500 kyr BP in Europe (Thieme 1997) and 790 kyr BP in the Near East (Goren-Inbar et al. 2004). Analyses of micro-charcoal in deep-sea cores off Iberia and France indicate that from 70 Kyr to 10 kyr BP fire regime followed climatic variability and subsequent effects of climate on vegetation (Daniau et al. 2010). Warm and moist periods were correlated with the development of Mediterranean forest and scrub communities and high fire frequency regime, whereas cold and dry periods were followed by steppic vegetation and low fire frequency regime. There is no evidence that large-scale alterations of fire regime were caused by humans (Neantherlands or modern) up to 10 kyr BP (Daniau et al. 2010).

There are numerous evidences from sedimentary deposits covering the Late Quaternary through the Holocene that indicate that extant species thrived in an environment of fire. The fire regime emerging from some of the best characterized sequences in south Spain is one of long fire-return interval (300-400 yrs) that increased as the climate became warmer and drier (150 yr), but the fire regime still staying as one of a long fire-cycle (Carrion et al. 2003). A pattern of low/high fire frequency associated to cold/warm moist/dry conditions is also present in other areas around the Mediterranean during the earlier part of the Holocene (Vanniere et al. 2008). Similar long fire cycles were also characteristic in the northern, boreal forest (Carcaillet et al. 2007), or in central Europe (Clark et al. 1989).

In southern Iberia the minimum fire activity was reached by the mid-Holocene (Gil-Romera et al. 2010). Since then, fire frequency became much higher, as human populations expanded. Here and elsewhere much of the increase in biomass burning since this period (6000 BP) can be attributed to cultural development (Carcaillet et al. 2002). Increased human activity through fire was accompanied by changes in the dominant vegetation: expansion of sclerophyllous trees, opening of the landscape with incorporation of scrubs, and abundance of xeric vegetation and grasses in more xeric areas (Gil-Romera et al. 2010). Note that climate change itself has produced significant vegetation changes in the past, independent of fire (e.g., favouring evergreen oaks at the expense of deciduous oak), although increased fire frequency not necessarily always favoured evergreen oaks (Colombaroli et al. 2009). By contrary, the expansion of some tree species, like *Fagus sylvatica* or *Pinus sylvestris*, following changes in climate was probably halted due to human actions driven by fire (Peñalba 1994, Figueiral and Carcaillet 2005).

Abrupt changes in vegetation have been documented when fire frequencies reached 30-50 yr (Carrion et al. 2001). The increased fire frequency and the associated changes in vegetation in favour of shrublands, together with higher presence of ruderal species, indicate that fires were mainly linked to the management of the territory (Carrion et al. 2003). Man use of the territory through historical times has involved the use of fire, grazing, ploughing and coppicing, among other. By the turn of the first millennium forest retrogression was near its maximum (Mugica et al. 1998). Since then, and until the middle of the XX century, the occupancy of the territory remained at its highest. Therefore, in SEM extant landscapes and vegetation composition are the legacy of centuries of intensive use of the land (Naveh 1994).

3. RECENT TRENDS IN FOREST FIRES

3.1 Trends in landscapes and interactions with fire

By the second half of the XX century the process of extensive land use is halted and reversed in most of Europe. Rural exodus, mechanization of agriculture, reduced extensive grazing, forestation in many marginal areas and changes in life styles became the norm (MacDonald et al. 2000). All of this introduced important changes in land-use and land-cover (Lehouerou 1992). Lan(Lloret et al. 2002)dscapes poor in tree cover, or in vegetative cover in general, gave raise to others that little by little accumulated vegetation, trees or shrubs, and in which management was less active. In addition, afforestation, often with highly flammable conifers was frequent (Pausas and Verdu 2008). This tendency to increase forest and wildland, that is, of adding fuels to the land, has been the norm in most countries (Fig. 1).

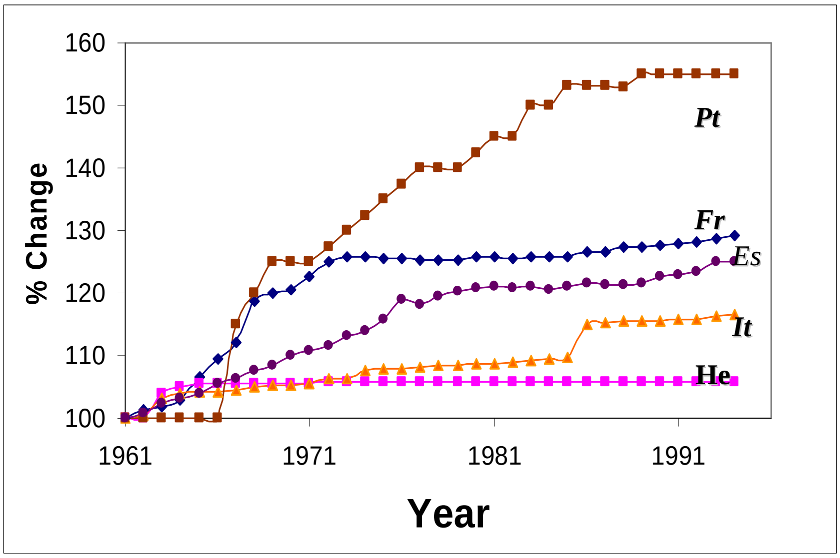


Fig. 1. Change (%) in forest area in the some Mediterranean countries of Europe during the second half of the XX century. Source: FAOSTAT.

Fires started to become more frequent as the vegetation (i.e., fuel) accumulated. Indeed, a significant relationship between fuel accumulation rate and fire has been found in some countries (Rego 1992). Landscapes became more homogeneous and highly interconnected (Peroni et al. 2000, Moreira et al. 2001, Lloret et al. 2002, Mouillot et al. 2005). This possibly contributed to enhance fire risk, provided that discontinuities are important to deter fires (Vega-Garcia and Chuvieco 2006, Viedma et al. 2009). Fires themselves may have further added to this process of homogenization (Viedma et al. 2006, Loepfe et al. 2010). Further, it has been shown that in some forest areas once an area burns it tended to reburn at an accelerated rate (Vázquez and Moreno 2001, Salvador et al. 2005).

Fire is major agent in structuring the landscape. In fire-driven landscapes, landscape structure is the result of fire regime, and changes in fire regime are reflected in landscape configuration (Johnson and Van Wagner 1985). Fire, therefore, is an agent of heterogeneity in the landscape. In SEM, however, fire is playing a dual role. There are many areas that are in various states of succession after abandonment. The initial fine-grain mosaic of patches derived from extensive use of the land is being lost as succession follows. Shrubland invasion is much quicker than that of forest (Mouillot et al. 2005). Fires can deter forest colonization in favour of shrublands (Trabaud and Galtie 1996, Mouillot et al. 2005, Acacio et al. 2009), erase previous patterns due to the development of vegetation of similar structure, regardless of the recent land-use history (Pérez et al. 2003, Duguy and Vallejo 2008), and increase fire hazard due to the homogeneity of the landscape that they favour after fire (Viedma et al. 2006). Consequently, fires can quickly erase the old, fine grain landscape pattern from the previous use of the land. Landscape patterns in the future may tend to be controlled by fire, changing the finer grain landscape to a coarser one depending on the fire-size frequency distribution of each area (Fig. 2; Fig. 13).

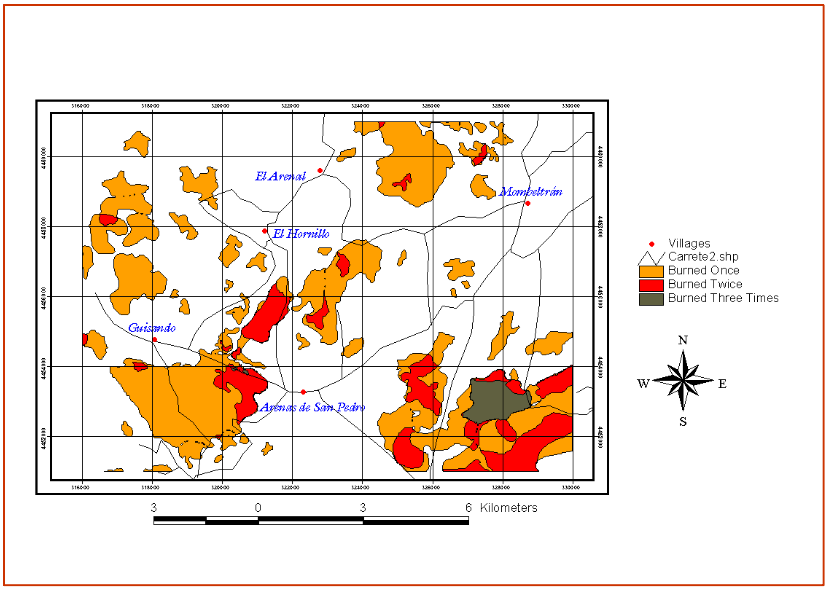


Fig. 2. Fires dominate the landscape in many areas across southern Europe and the MR. Here is an example of the southern Sierra de Gredos, in Central Spain. Maps are the fires occurred from 1970 to 1990. Note the variety of sizes and shapes of fire perimeters and that some areas have burned repeatedly. From (Viedma et al. 2006).

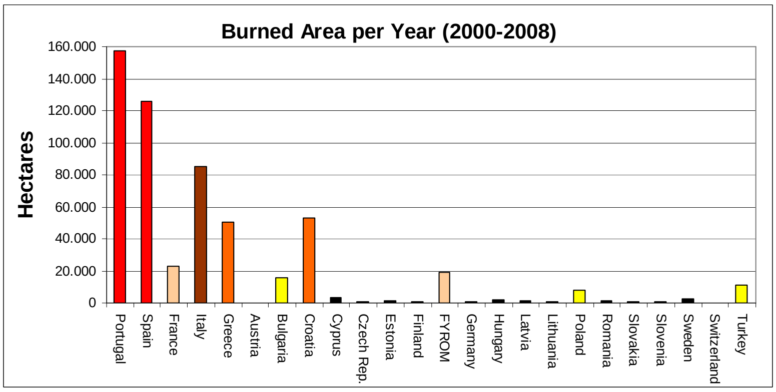
Large fires can play a significant role in this process of homogenization since large areas will tend to be tuned to the time since last fire by way of being regenerating after fire. Large fires, however, may produce some heterogeneity as well due to the fact that they propagate during days or weeks, under different conditions and affect different land-cover and vegetations types and topographies. The degree of heterogeneity can vary depending on the conditions of fire. It has been found that high severity burned areas can vary from ca. 50% to more than 85% when conditions are severe (Oliveras et al. 2009). That means that even though the majority of a large fire will burn with similar severity, partially burned or unburned islands may not be uncommon. These islands can play a critical role in the colonization of both plants and animals (Ordoñez et al. 2005, Moneglia et al. 2009).

3.2 Trends in fire occurrence

Although the forest fire statistics are incomplete for the first part of the XX century, the available data indicates that wild-fires were not important until the middle of the century, at least in the forested areas, which were the ones for which statistics were compiled. By the late 1960's wildfires started to occur at an increasing rate in the Mediterranean countries of the EC (Alexandrian and Esnault 1998). The number of fires has continued rising, although part of it is due to a change in the compilation of statistics. Area burned, which is less sensitive to compilation procedures, increased during the 1970's and into the 1980's, by which time Spain and Italy had reached maximum values. Greece and Portugal followed suit with some delay (Moreno et al. 1998). It is important to note that the increase in fire occurrence during the 1980's did not correspond with changes in the mean climate, which were different in the East and West sides of the Mediterranean (Anonymous 1995b). Additionally, an increase in ignition sources due to changes in socioeconomic and land-use cannot be excluded. Therefore, while a climate effect cannot be ruled out, certainly other factors came into play.

During the last decades Spain, France and Italy have decreased the area burned, but that is not the case for Greece and, much more so, for Portugal or other Balkan countries, that have increased very markedly the area burned per year (Fig. 3). During the transition towards high fire incidence, from the 1960's into the 1980's, other countries in southern Europe and Asia with a Mediterranean climate (e.g., Turkey) did not experience the same process. Fires are also occurring in other parts of Central and northern Europe, although with a much lower significance (Fig. 3). Although these numbers must be taken with caution provided that statistics are not consistent across countries, the magnitude of the difference suggests that by the middle of the last century different processes were going on in different countries and development was an important factor (Vélez 2009). Using the decade of the 1980's as reference, the significance of the area burned in the various countries is such that fire rotation periods vary from about half a century (Portugal [42 years], Italy [53]), to a century (France [83], Spain [110] and Greece [110]) and many centuries (Turkey [1670] as well as in the other central and northern European countries (Moreno et al. 1998). These contrasted patterns of burning have continued into the 1990's (Goldammer 2002).

Fires became more frequent during the second half of the XX century, but also more widespread. The analysis of change in the geographic distribution of fires in Spain illustrates this process: early in the 1970's fires were more restricted to areas were forest dominated. With time, fires became widespread throughout the whole country (Moreno et al. 1998, Mugica et al. 1998) (Fig. 4).



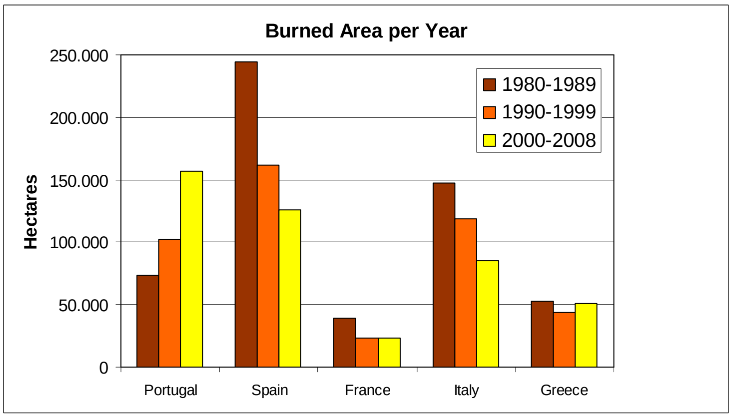


Fig. 3. Area burned per year in a number of European countries (upper panel), and change in total area burned per year during the last three decades in some Mediterranean countries of Europe (lower panel). Note that the differences in forest land is large among countries, hence the relative importance of fire may not correspond with absolute figures. Source: JRC Forest Fires Statistics.

Around the Mediterranean most (≈95%) fires are caused by humans, either by accidents or intentional (San-Miguel-Ayanz et al. 2009). A small percentage of fires are caused by lightning (e.g., in Spain just 4%), although they tend to cause larger impacts (e.g., in Spain around 10% of the burned area), because they usually occur in remote forest areas, where the detection and first attack takes longer. In addition, they are frequently associated to extreme meteorological conditions, with dry thunderstorms and strong winds, which sometimes make difficult the use of aerial extinction resources. In general, the length of the fire season and the temporal pattern of fire occurrence vary across regions and with type of ignition (negligence, arson, agriculture-related activities or lightning). Lightning-ignited fires tend to concentrate almost exclusively in the summer while fires related to pasture burning are more widespread during the year, well into spring and fall, accidental fires being somewhere in between. These patterns can vary from one region to another. Fires vary also in their location within an area, and in geographic distribution depending on the source of ignition.

Fires are extreme phenomena, the result being that few fires account for a very large percentage (2.6% of fires larger than 50 ha account for 76% of the area burned in Southern Europe) (San-Miguel-Ayanz et al. 2009). This varies across regions, with more moist/cool regions being less extreme than dry/warmer ones (Ricotta et al. 2001). The degree of extreme-fire varies also at a given area between years. In this case, more cool/moist areas and warm/dry areas vary less than areas with a more contrasted climate (Vázquez and Moreno 1995).

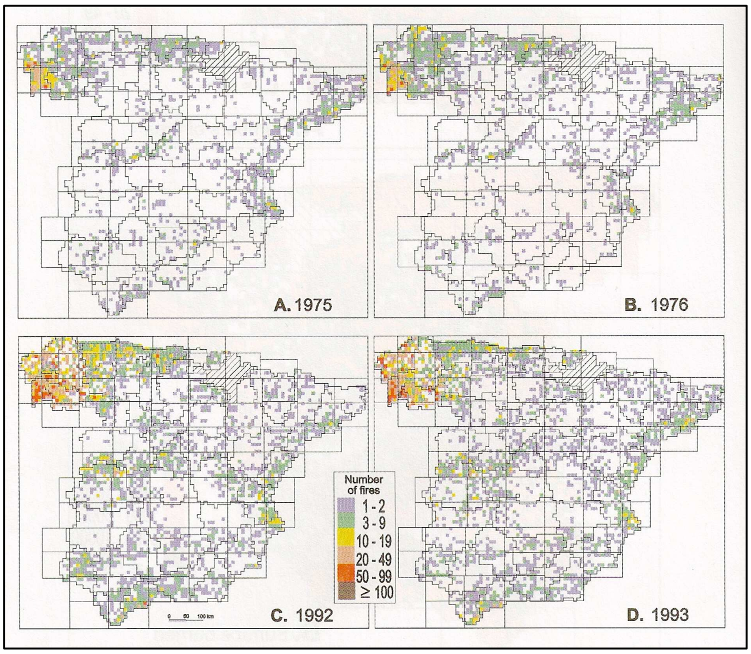


Fig. 4. Number of fires per grid-cell (10x10 km) in Spain during: A) 1975; B) 1976; C) 1992; D) 1993. Although a similar area was burned during these years, the geographic distribution changed notably. From (Moreno et al. 1998).

All fires are suppressed, and this policy has been maintained until now in most of Europe. As fires started to occur, fire fighting capacity of EU Mediterranean countries became a priority, and little by little fire fighting capacity was built up. Each summer more than 400 aircraft provide support for fire suppression in the EU Mediterranean countries. Expenditures in prevention and suppression amount to more than 2.5 billion € per year. Of this, 60% goes to suppression equipment, personnel and operations, the rest being used for preventive tasks (Vélez 2008). A reflection of this is increment in fire fighting capacity is that, on average, there has been a tendency to decrease the mean size of fires. Although, in general, the number of large fires seems stable (San-Miguel-Ayanz et al. 2009), in some areas is increasing (Gonzalez and Pukkala 2007).

3.3 Ignitions and human influence

Since most ignitions are of human origin, and not all areas in a landscape are likely to have similar human influence, an important element for understanding fires is what factors control where they break out and, depending on that, which areas in the landscape are most at risk. Studies have shown that distance to roads or towns are significant variables in determining fire risk (Badia-Perpinya and Pallares-Barbera 2006, Romero-Calcerrada et al. 2008, Wittenberg and Malkinson 2009). Depending on the point of ignition and on fire propagation the effects are such that, contrary to what may be believed, areas at intermediate distance to towns or roads might burn in higher proportion than those closer. Therefore, the wildland-urban interface (WUI) (Lampin-Maillet et al. 2010) becomes an important element in determining the risk of fire. The WUI adds additional risk provided the increase in secondary housing along the coasts and in the uplands in the vicinity of large conurbations. Land abandonment in the surrounding of small villages and farms, where natural vegetation is invading the old fields and close to the houses, adds additional risks. Proximity to the WUI interface and roads is a factor of risk that needs to be properly weighted in fire-prone protected areas.

3.4 What fires burn?

The fact that current fires are overwhelmingly driven by man has some important consequences for understanding what areas and vegetation types are being burned. Contrary to what might be expected, fire incidence is not highest at the places were climate-driven fire danger is highest. For instance, in some of the more mesic areas of the Iberian Peninsula, where extant vegetation is not of Mediterranean type, fire-occurrence is at its highest (Vazquez et al. 2002). Since people cause fire, they select where and when these occur and the temporal patterns can be highly predictable (Diez et al. 1994). Thus, in favorable (warmer, drier) years the fire season can be shorter, not longer (Vázquez and Moreno 1995), which contradicts normal expectations.

During the course of the century, the types of vegetation burned have been changing, from more wooded-dominated areas to shrubland-dominated areas. This may reflect in part the changes in land-use and land-cover experienced during this time. Fires burn through any area that contains sufficient fuel and that includes protected areas as well (Fig. 5 and 6). During the last years, of the total area burned by forest fires in southern Europe, nearly 30% were of Natura 2000 sites. Fires, however, do not burn at random the vegetation that is available (Nunes et al. 2005, Moreno et al. in review). Needle-leaf forests are burned preferentially across the Mediterranean (Syphard et al. 2009).

Fire occurrence may be linked to particular abiotic or human factors. In areas where lightning is the main source of ignition, fires preferentially occur at higher elevations and in places where tree vegetation is dominant. Fires also have preference for certain topographic locations, or distances to towns or roads (Mouillot et al. 2003, Badia-Perpinya and Pallares-Barbera 2006, Syphard et al. 2009). In general, remote, less densely populated, more distant and less accessible areas tend to burn less. This is important for conservation planning, and calls for a need of planning that is compatible with the conservation of these areas, if only by modifying urbanization patterns to reduce ignition sources. Furthermore, since it is likely that in the past the first areas to be abandoned were those more distant to towns and less fertile, that means that new abandonments will tend to occur closer to towns, were ignition sources are more abundant, and in more fertile areas, thence with higher productivity, which means that fire hazard and risk will increase.

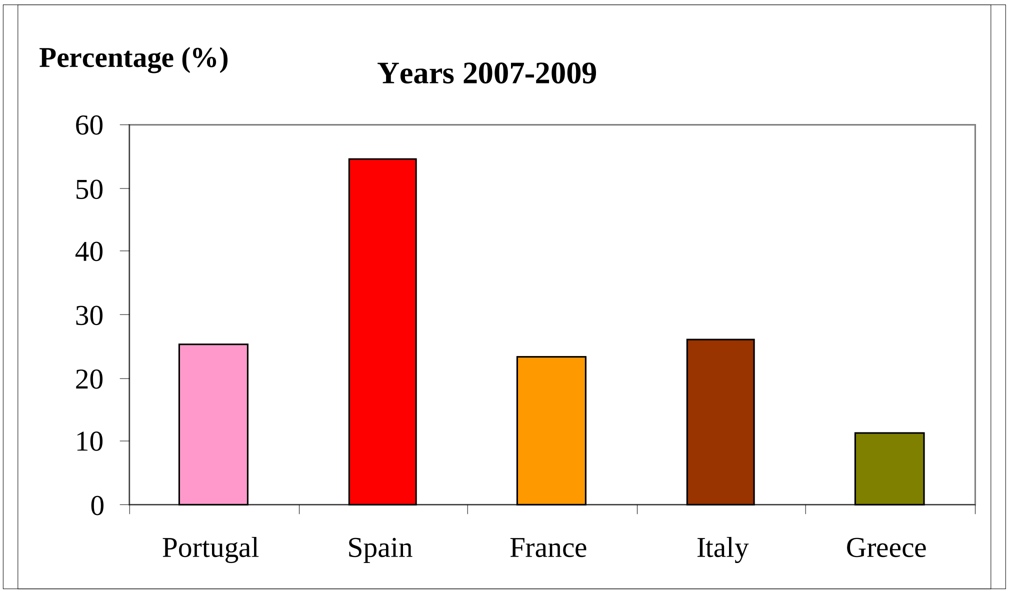


Fig. 5. Percentage area burned by forest fires that were Natura2000 sites in several EU countries during the years 2007-2009. Note that in some countries, more than half of the area burned can be protected areas. From (Camia et al. 2009).

The combination of afforestation with conifers close to towns in former cultivated relatively fertile areas seems the worst option.

4. FIRE, CLIMATE AND FIRE-REGIME

4.1 Fire weather and climate

The close relationship between weather, climate and fire has been attested in many places, whereby high temperatures and low precipitation play a most significant role, albeit in the case of rainfall this can be non linear and with lag effects. Few particularly severe weather days can account for a large percentage of the area burned (Vázquez and Moreno 1993, Piñol et al. 1998, Viegas 1998, Pausas 2004, Trigo et al. 2006). Interestingly, it has been found that the fire-size frequency structure matches that of the weather-severity frequency structure in several regions (Boer and Lambert 2008), which hints at a close local relationship between weather events and fires. Multiple, very large fires episodes, concomitant with most severe climate and weather extremes (drought and heat waves) are additional recent features of fires in SEM. Episodes such as the 1994 in the Spanish Levant (Moreno et al. 1998), of Portugal in 2003 (Trigo et al. 2006), and Greece in 2007 (Founda and Giannakopoulos 2009), with hundred of thousands of hectares burned, mark unprecedented records of how extreme fires can be in many regions. Under these conditions, fires can expand towards areas in which until now they were rare or absent (Camia et al. 2008) (Fig. 7).

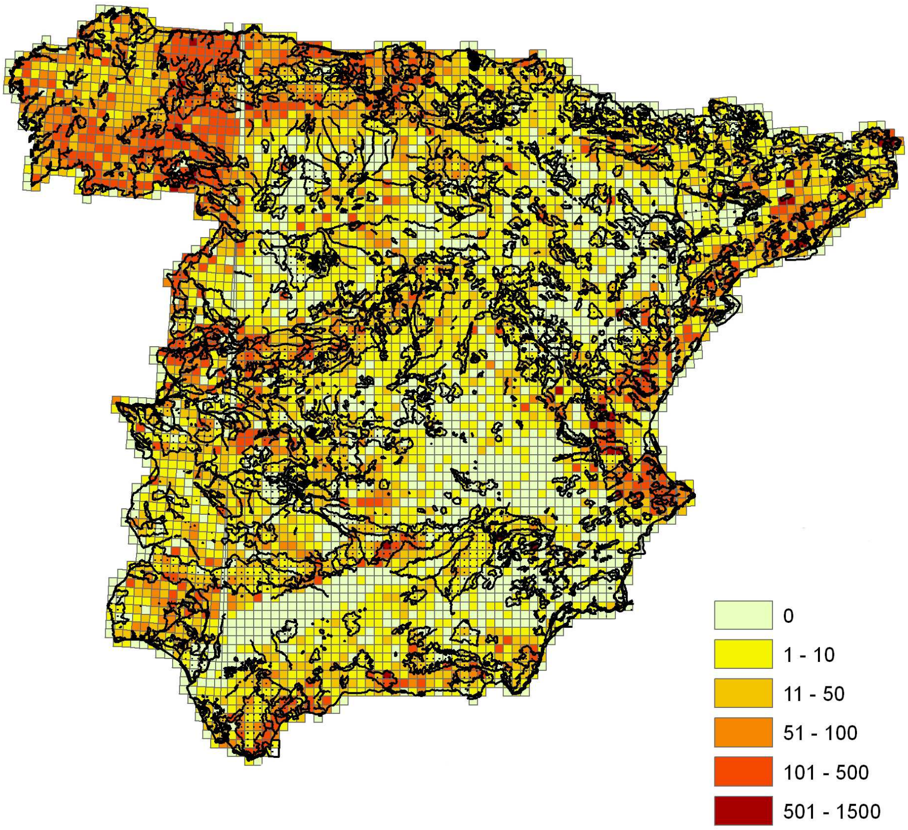


Fig. 6. Area burned per year per 10x10 km cells in Spain during the last 28 years superimposed to the natural protected areas, including the Natura2000 sites. Note that fires are occurring at or surrounding protected areas virtually all throughout the country. From: Rodríguez Urbieta, Zavala, Mateos and Moreno, in press).

4.2 Fire regimes and interactions with other stressors

Mediterranean ecosystems can be considered to have evolved under fire-dependency/influence (Pausas and Verdu 2005). Nevertheless, the fire regime under which extant species have persisted until these days is poorly known. There are evidences that, at least during the most recent two centuries, the pattern of burning was related to human activities that resulted in frequent (5-10 fires per century), low-intensity, surface fires that permitted the persistence of tree populations (Vega 2000, Fulé et al. 2008). The lack of correlation between drier periods and fire indicates that fires were deliberately lit to stay within certain bounds (Fulé et al. 2008). Burning patterns by shepherds in many other areas might have followed these patterns. In fact, these human-driven fires are the basis for the conservation of some areas in Europe (Goldammer and Page 2000).

Current fire regimes are, however, very different from any previous distant or recent fire regime. For instance, the geographical pattern of fire occurrence of fires caused by people or by lighting in Spain is very different (Vázquez and Moreno 1998), which suggest that current fire incidence is changed with respect to more natural patterns. Similarly, the geographical patterns of fires caused by pasture burning, which might be more similar to management-driven fire regime, is also very different from the current pattern of burning or to the natural one. Temporal patterns of occurrence are also very different depending on the ignition source (Vázquez and Moreno 1995). Lightning ignited fires tend to occur in a narrow window during the summer, whereas people caused fire are much more widespread, with fires related to pasture burning being shifted towards spring or fall. Consequently, geographical and temporal patterns of fire are most likely to differ from any previous recent or distant ones. The example of Spain can be extrapolated to other SEM countries where ignitions caused by people dominate (San-Miguel-Ayanz et al. 2009).

Fig. 7. Fires during the extreme event of 2007 in the Eastern Mediterranean. Note that fires extend beyond the Mediterranean coasts into the interior. From (Camia et al. 2008).

Changes in fire regime, in combination with other long-term anthropogenic disturbances, may cause further fire-induced degradation beyond the resilience domain of Mediterranean ecosystems. As a consequence of this long-term human impact, most of the Mediterranean basin is now regarded as 'degraded' . Therefore, fire impacts on ecosystems should be analyzed in terms of the interactions between direct fire-induced processes and previous human-induced degradation processes.

5. ORGANISMS RESPONSES

5.1 Plants

5.1.1 Mechanisms of fire responses

Post-fire regeneration in fire-dependent ecosystems, as in the MR, usually follows the autosuccessional process (Trabaud 1994). Plants are able to withstand fire by surviving it as adult, and resprouting shortly thereafter from various structures: aerial buds (*Quercus suber*), lignotubers (*Erica arborea*), or roots (*Quercus coccifera*), among other. By contrary, the living individuals of other species succumb to fire, and their regeneration depends on seeds stored in the canopy (*Pinus halepensis*), or in persistent soil seed-banks (*Cistus ladanifer*). Fire directly (heat exposure, smoke) (Trabaud and Oustric 1989, Pérez-Fernández and Rodríguez-Echeverría 2003, Luna et al. 2007, Moreira et al. 2010), or indirectly (changes in light, pH, Nitrogen) (González-Rabanal and Casal 1995, Henig-Sever et al. 2000, Luna and Moreno 2009) releases the seeds from dormancy and promotes their germination. Consequently, post-fire plant dynamics rests largely on endogenous sources. After fire there is usually a flush of germination of all plants, including a rich layer of herbaceous species. Species richness is highest a few 3-5 years after fire. Thereafter, competition among shrubs and trees displaces lower stature plants, the structure becomes more complex and the whole system follows a dynamic that in a few years will render a similar composition and structure to that before fire (Trabaud 1994). Long term studies in *Quercus ilex* stands in southern France show that the system should be stable to ca. 50 years fire cycles (Jacquet and Prodon 2009).

This direct regeneration model, however, may not always apply. For instance, there are several tree species that regenerate poorly or not at all after a wildfire, and may be threatened by fire. Some documented examples include *Pinus nigra* and *Pinus sylvestris* (Herranz et al. 1996, Riera and Castell 1997, Luis-Calabuig et al. 2002, Retana et al. 2002), *Pinus pinea* (Rodrigo et al. 2007), *Abies pinsapo* (Cabezudo et al. 1995), *Abies cephalonica* (M. Arianoutsou, pers. com.), *Fagus sylvatica* (Herranz et al. 1996); *Juniperus oxycedrus* in western Central Spain (J.M. Moreno, personal observation), *Juniperus phoenicia* and *Juniperus communis* (Quevedo et al. 2007), *Juniperus thurifera* (Gauquelin et al. 1999). A recent study (Paula et al. 2009) provides a comprehensive list of species (952 taxa) with regeneration traits, many of which are considered fire-sensitive, that is, they do not resprout or generate from seeds after fire. Some of these forests are very important from a conservation point of view, and are part of protected areas, including Natura 2000 sites. Direct regeneration, that is, the recovery of the system after fire to its prefire situation was found to have less than 50% chance in nearly half of the tree species analyzed in Eastern Spain following a very large fire (Rodrigo et al. 2004). Consequently, fire in the MR can introduce changes in vegetation that are likely to be long lasting, until the fire sensitive plants are able to colonize again the burned area.

Unfortunately, until now little research has been carried out for establishing recolonization rates of fire sensitive species, and the few done have been restricted to some key tree species. For instance, *Pinus nigra* is able to recruit from unburned individuals or individuals that survive the blaze from less than 100 m (Ordoñez et al. 2006), and something similar is for Abies babilonia (Arianoutsou, pers. com). Other species that have dispersal mediated by birds can reach several kilometers (acorns and jays) (Pons and Pausas 2007). Little is known about colonization distances and rates of shrubs or other low-stature plants that may have low regeneration capacity.

5.1.2 Fire regime and plant responses

Even in the case of species that have mechanisms to withstand fire and regenerate well thereafter, there are a number of mechanisms that can promote changes in plant populations and species. These include fire-interval effects or event-dependent effects (fire intensity, fire season, climatology after fire, etc.) among the most important.

5.1.2.1 Fire interval

Fire driven vegetation is highly resilient to fire as long as fire return interval is not too short to allow them to recover from a previous fire and replenish regenerative capacity. Since most fires occur under severe conditions, it cannot be ruled out that fire will propagate through stands even in their young stages of recovery, hence even with reduced biomass (Trabaud and Galtie 1996, Vázquez and Moreno 2001). Consequently, fire can revisit an area when plants have not reached reproductive maturity, and seed banks and resprouting capacity may have not been fully restored. Since age-to-maturity is dependent on plant size, trees may have greater difficulties than shrubs in an environment of increased fire frequency. Time to maturity in *Pinus* species is 10+ years. When fires reburn at short intervals *Pinus pinaster* or *Pinus halepensis* stands, the new fire will turn them into shrublands (Vallejo and Alloza 1998, Pérez et al. 2003, Espelta et al. 2008), the fate of which will be uncertain and depend on future burning. Modeling studies show that increased fire frequency augmented seeder species with short time interval for maturity, reducing tree abundance. Resprouters were less affected by this since they usually have large stores of reserves (Cruz et al. 2003). Plant functional types tended to decreased with increasing fire frequency (Pausas and Lloret 2007).

A question of interest is to know whether positive loops in fire occur. Indeed, it has been observed that, in some areas, reburning tended to occur at a much shorter interval than when burning for the first time (Vázquez and Moreno 2001, Salvador et al. 2005). Furthermore, areas with high recurrence of fire had reduced vegetation growth (Diaz-Delgado et al. 2002, Delitti et al. 2005), possibly due to nutrients being lost at a higher rate than they were replenished during the short fire cycles. Reduced vegetation growth can imply a lengthening of the period during which erosion risk will be high due to lack of vegetative cover (Francis and Thornes 1990, Vallejo and Alloza 1998). Post fire erosion rates very enormously between sites and years, but can be very large (Pausas and Verdu 2008). High soil erosion rates in impoverished soils can be irreversible at the ecological time scale. Therefore, an acceleration of the fire cycle can change vegetation rapidly, alter nutrient pools and reduce soil fertility, all of which can defer for long the recovery process of the vegetation towards more mature stages.

Of particular interest are the many areas that were cultivated until recently but abandoned. Many of such old-fields are colonized by opportunistic species that are seeders (*Ulex*, *Genista*, *Cytisus*, etc.) (Casal 1987, Gallego Fernández et al. 2004, Baeza and Vallejo 2006). These species have high growth rates, which combined with the relatively fertile soils of abandoned lands allow them to accumulate large quantities of fuels, particularly dead-fuel, in a short time, making them particularly prone to fire (Pérez and Moreno 1998b, Baeza and Vallejo 2006). Fire-prone shrublands that have colonized old fields often enter into short-interval fire cycles that stop any further secondary succession towards more mature vegetation. Repeated fire expose them to further erosion and degradation for long periods of time (Vallejo and Alloza 1998). Therefore, land abandonment facilitates fire cycles that result in ecosystem degradation loops, rather than in a directional process of recovery towards more mature states (Dobson et al. 1997). Increasing ecosystem resilience capacity in these abandoned lands would thus require breaking such loops and promoting secondary succession towards more mature, more resilient vegetation (Vallejo and Alloza 1998).

5.1.2.2 Event dependent effects

Fire intensity and severity

Fires vary enormously in the energy they release, and this is a function of fuel quantity and quality and meteorological conditions during fire. High fuel loads, particularly dead-fuel, and high temperatures and low humidity conditions usually produce higher fire intensity than lower fuel loads or less severe conditions. Because climatological and meteorological conditions are so important in the fire process, inferring fire intensity and subsequent severity from fuels is not always appropriate (Bessie and Johnson 1995). Different elements (vegetation type, topography) can produce different levels of fire-intensity and severity depending on how fire propagates (Oliveras and Bell 2008). Measuring fire intensity, or surrogate measures of it, such as temperatures, is normally not possible during wildfires. During experimental fires, thermocouples or other devices have been used to record temperatures or infer energy release or other measures of fire intensity. After fire, surrogate, measures of the impact of fire, such as crown damage, height of scorch, ash type, etc. have been use (Keeley 2009). In addition, measuring the minimum diameter of the burned branches (Moreno and Oechel 1989), which is related to temperature residence time (Pérez and Moreno 1998b), has proved useful in various tree (Gill and Moore 1998) or shrubby systems (Keeley et al. 2008), permitting measurements on large scale. In addition, fire severity or burn severity can be mapped based on field recognisances or by using satellites, hence permitting mapping this variable at large scales (Brewer et al. 2005).

Experimental fires usually demonstrate that as fire intensity increases, plant responses decreases, although the relationship is not linear for all plants (Moreno and Oechel 1994). High intensity fires produce higher mortality of shrubs and trees (Pausas et al. 2003, Broncano and Retana 2004). Seedling germination, establishment and growth will be variable in relation to fire severity, depending on rates of seed dispersal by serotinuous plants, the amount of litter remaining after fire, elevation, dominant vegetation and soil fertility (Pérez and Moreno 1998, Pausas et al. 2003, Vega et al. 2008, Rudiger et al. 2010). While regeneration of the vegetation may be lower the higher the fire severity (Diaz-Delgado et al. 2002) the dynamics after fire can rapidly homogenize burned areas (Pérez and Moreno 1998). Until now, no studies are available from the MR across a whole fire cycle that could illustrate the long-term effect of changes in fire severity. Studies from other Mediterranean environments indicate that carry over effects are detectable, and that positive loops may occur. That is, places burned with at higher severities may later reburn at higher severities as well (Thompson et al. 2007). Other studies in shrublands, however, have found negative loops (Moreno et al. in review). Therefore, it is still premature to evaluate how the likely change in fire intensity and subsequent severity on plants in the southern Europe and the MR may be influencing extant and future vegetation and landscapes.

Fire size

Fire size can affect plant population and community changes to the extent that fire sensitive species will have to colonize the burned area from the surroundings or from unburned islands. Large fires may occur under more severe weather, which is likely to cause increased fire intensity and impact on plants, adding further strain on the affected populations. Distance to the unburned edge was important for the recovery of some key tree species that are fire sensitive, as is the case for *Pinus nigra*. Unburned islands were also an important source of seeds (Retana et al. 2002). Little is known about the relationships between fire size, distance to the edge and plant dynamics after fire for other functional groups of plants other than trees dispersed by wind. In the case of herbs, no relationship has been found for herb species richness and distance to the fire-edge in the first few years after fire (Moreno and Cruz 2000).

Fire season

Fire season can interact with fire, among other, by the changes in the seed bank or in plant physiological status that occur during the year in relation to penology and that may affect resprouting potential. Soil seed banks in Mediterranean ecosystems are known to persist through time, although its fluctuation from year to year and within the season has not yet been well quantified (Valbuena et al. 2000, Clemente et al. 2007). It can be argued that the size of the seed-bank will change during the course of the season, being highest at the time of maximum dispersal. Therefore, the size of the soil seed-bank at the time of fire could vary depending on when the fire occurs, and this could affect the number of germinates and subsequent recruits. In the MR, most species disperse their seeds in the summer (Bastida and Talavera 2002, Pérez Latorre et al. 2007), although some species, like *Cistus*, can disperse them well over the winter season and some (*Rosmarinus officinalis*) in spring (Moreno and Cruz 2000). There is not a clear relationship between dispersal pattern and the period of natural occurrence of fires, which would occur in the summer. In a year, seed rainfall can be enormous and very variable among species, from 700 seeds m^{-2} for *R. officinalis*, to 7000 seeds m^{-2} for *Cistus ladanifer* to more than 100.000 seeds/m^{-2} for *Erica australis* (Moreno and Cruz 2000, Moreno et al. 2004). Despite this, no differences in recruitment were found among fires conducted in early or late season (Moreno and Cruz 2000). Differences among *Cistus* and *Rosmarinus* for total seedlings were almost one order of magnitude larger in the first than in the second, and in any case well above the numbers needed to replenish both populations after the burn. Survival was higher for *Rosmarinus* than for *Cistus* (Quintana et al. 2004), which indicates that other mechanisms could compensate germination differences. Therefore, it is unclear yet how seed rain and seed bank dynamics affect plant regeneration potential after fires in different seasons.

As for resprouting, fire season was an important factor in determining resprouts density of *Quercus coccifera* in southern France, whereby early season burning produced greater response than late season burning (Malanson and Trabaud 1988). Studies in central Spain (Moreno et al. 2004) showed that this might be true as well for some species (*Phillyrea angustifolia*), although differences were not very important, but not for others (*Erica australis*). In this case, early season fires were more detrimental for smaller individuals. Large size individuals showed no differences in their survival in relation to fire season. Resprouting growth was higher in early season burns than in late season ones, similar to that reported from Greece for *Arbutus unedo* (Konstantinidis et al. 2006). The effect is such that early season fires can produce resprouts within the same season and late season fires can delay resprouting growth several months, causing a delay in regeneration of almost one season. Consequently, fire season can differentially affect plant populations depending on the species and in relation to the size of the plants.

Fire climatology

The effects of climate variability on postfire plant dynamics deserve particular attention. Mediterranean climates are characterized by warm and dry summers, and cool and moist winters. Rainfall is very variable from year to year, and so it is the length of the rainy season. The coefficient of variability of rainfall increases with decreasing precipitation (Martin-Vide and Lopez-Bustins 2006), which means that the lower the precipitation in a given area the more variable it is through time. Furthermore, droughts are common. Fires, particularly very large fires, and large fire-episodes tend to occur under a few summer days, with particular circulations patterns, long dry periods and heat waves (Pereira et al. 2005, Trigo et al. 2006, Founda and Giannakopoulos 2009). Therefore, post-fire regenerating environments can be subject to a large variability in rainfall patterns and rainfall extremes, among other climatic conditions. In the case of large fires or large fire-episodes, that means that large extensions that can amount to significant portions of a country will be regenerating and subject to similar climate conditions. For instance, in 1994, during the first week of July, funnelled by a heat wave and a protracted drought, all along the whole Spanish Mediterranean coast, from virtually the French border to Cádiz, 17 very large (>1000 ha) fires were burning, causing a total of 170 kha. This was the first very large fire episode recorded in southern Europe. Additional episodes followed later in the summer, affecting a total of more than 400 kha. In the Region of Valencia, these fires burned 14% of the forest territory (Anonymous 1995a). In Portugal, in 2003 and again in 2005, two very large fire episodes funnelled by heat waves burned >400 ha and >300 ha, respectively. These amounted to 12% and 8% of the forest area of the country. In Greece, similar meteorological conditions burned 316 kha in 2001 and 270 ka in 2007, amounting to 9% and 7%, respectively, of the forest area of the country. Therefore, the effects of climate variation on plant populations under regeneration could be geographically extensive and, eventually, lead to long-lasting changes.

Fire-recruiter species are a main component of the Mediterranean. These are species whose individuals die during the fire, the recovery of the population resting entirely on postfire germination and establishment. Therefore, they can be particularly sensitive to the vagaries of weather and climate after fire. Germination is commonly triggered by the first rains (Daskalakou and Thanos 2004, Quintana et al. 2004). However, the optimum germination temperature varies among species (Galmés et al. 2006). Therefore, the temporal pattern of rainfall can shift up to several months the time of germination depending on when rainfall occurs and the temperature reigning at that time. Once germination has occurred, soil moisture is probably a most limiting factor for survival. Mortality during the first few months, up to and including the first summer, is very high (Quintana et al. 2004). Consequently, the climatology following fire can play a significant role in triggering germination and causing mortality in fire-recruiter species. Empirical studies show that plant persistence can be at risk in the case of severe climatology following fire, as in drought (Quintana et al. 2004). This study showed that *Erica umbellata* regeneration was at the level of virtual local extinction due to a severe drought following fire. Not all species were equally sensitive to drought, a fact that could produce shifts in relative dominance of them. Furthermore, rainfall patterns, by altering the germination temporal patterns can determine survival and fitness (De Luis et al. 2008). A differential sensitivity among coexisting species towards the postfire climate, either for germination or for survival, could determine the future structure and composition of the plant populations and communities.

5.2 Animals

5.2.1 Faunal general responses

Fire temperatures will reach levels that will be lethal for all organisms if directly exposed to the passage of fire. Animals may flee away or seek refuge underground or inside trunks and survive the fire, particularly in areas where fire intensity might be reduced. Direct killing of animals of certain size and mobility is not common, although when it occurs it makes the headlines. This is more likely to occur in animals with slow mobility and/or limited access to safe sites. Few examples exist in the literature that a population might have gone extinct as a result of fire (Lyon et al. 2000). However, the direct effect of fire on some endangered animal populations might be important. For instance, the Mediterranean turtles have been found to have very high mortalities as a result of fire (Stubbs et al. 1985, Popgeorgiev 2008). When adult, sexable individuals die the population recovery can take decades (Hailey 2000). When fire is combined with additional negative actions, the effect of the two factors can be most severe and decades will be necessary for the recovery of the population (Hailey 2000).

Most faunal studies usually address faunal population and community changes by means of short-term diachronic approaches complemented with synchronic approaches, by comparing recent burned sites with adjacent long-unburned sites. The dynamics that emerge from most such studies is compatible with an autosuccessional dynamic. Faunal dynamics appear to be closely related to the changes in the plant structure and composition as these affect the environmental conditions (more or less open habitat) or the resources (Prodon et al. 1987). The general pattern is that early stages of postfire succession are usually dominated by more opportunistic species that either survive the fire (refugia hypothesis) or populate the area from nearby areas (immigration hypothesis). Open habitat, more xeric species dominate the early stages after fire, with species proper of more mature stages coming at later stages. The initial changes in the community and populations induced by fire can be visualized by the change in rank-abundance diagrams models that before fire resembles a broken stick type and after fire takes the form of geometric model, indicative of a few species being more dominant (Fattorini 2010). As time passes and the plant community develops a succession with replacement of species follows.

Despite these generalities there are a number of variations around this model depending on animal characteristics. For instance, in a study of a *Quercus ilex* stand in southern France (Jacquet and Prodon 2009) during a 28-yr period it was followed at the same time plants and birds communities. Plant followed an autosuccessional pattern as described earlier. Animals (birds), however, did not follow a clear trend in species richness through time. Nevertheless, they found a replacement among the various guilds, whereby after fire granivorous species dominated, later granivorous-insectivorous, later insectivorous and finally frugivorous-insectivorous. The pattern of changes was coincident with that of vegetation composition and structure. They calculated that after 50 years both, plants and birds, would have reached the conditions prior to fire. Consequently, a 50 fire cycle would produce a stable system. Few other studies have followed faunal populations in SEM during such a long period and at the same site. In a study in Sierra de Prades (Eastern Spain) in a *Quercus ilex* stand, the micro-arthropods community of the soil declined after fire but densities recovered quickly (2-3years) after fire. Nevertheless, the community, particularly the Collembola, 15 years after fire had not still recovered to similar prefire levels (Mateos et al. 2005). These two examples covering very different groups of organisms indicate that faunal recovery times can be long, and diachronic studies are critical to avoid some of the assumptions of synchronic studies.

The time to reach a state similar to the one prior to fire is an important parameter since fires before that moment can have long-lasting effects on the community. Time to recovery varies depending on organisms and system. For instance, soil dwelling micro-invertebrates very quickly resembled that of prefire conditions in a phrygrana ecosystem of Greece, but not so those inhabiting in the litter (Sgardelis et al. 1995). Among soil dwelling arthropods in a *Quercus suber* stand of sourthern France, Prodon et al. (1987) found that groups feeding on living material, more dependent on the recovery of the litter layer and on transfer of energy across trophic links, required more than 20 years to recover their populations. By contrary, groups feeding on dead plant material or fungal hyphae required 10 years or less. Within a given group of organisms, the type of habitat may determine the rate of recovery. For example, ant populations recovered less rapidly in moist forest than in xeric ones (Arnan et al. 2006). Consequently, the postfire dynamics of faunal communities can vary depending on the group concerned, the niche they occupy, and habitat type. Groups and habitats with longer requirement to re-establish after fire are likely to suffer most from changes in fire regime that imply an increase fire frequency.

5.2.2 Faunal responses and fire regime

Animal population and communities are equally sensitive to variations in fire regime parameters. Following are some examples that illustrate how changes in fire regime can affect these organisms.

5.2.2.1 Fire return interval

The dynamics to recovery of abandoned lands towards forest will be hampered by fire. The resultant landscape will be one where areas of low and high fire frequency can coexist (Mouillot et al. 2003). Bird-diversity in Central Portugal decreased from forest to shrublands (Moreira and Russo 2007). As fires tend to transform during the initial stages forest to shrublands, scenarios of increased fire frequency would lead to reduced bird diversity in the area. Increased fire frequency would reduce other vertebrate communities in the Mediterranean (Moreira and Russo 2007).

In the Southern Alps fire have been a recurrent phenomenon and have usually occurred in winter and spring, affecting *Castannea sativa* forest. Arthropods populations were more resilient to single fires than to repeated events, recovering 6–14 years after a single fire, but only 17–24 years after the last of several fires (Moretti et al. 2006). Flying zoophagous and phytophagous arthropods were the most resilient groups. Pollinophagous and epigaeic zoophagous species showed intermediate resilience, while ground-litter saprophagous and saproxylophagous arthropods clearly displayed the lowest resilience to fire. Their species composition 17– 24 years post-burn after repeated fires still differed markedly from that of the unburned control plots.

5.2.2.2 Fire size

Animals with limited mobility are thought to be most vulnerable to fire and to the subsequent shelter loss. Recolonization rate depends on the ability of new animals to colonize the burned area from the edges of fire or from unburned islands or refugia. Establishing the source of the recolonization is important given the tendency towards large fires. Among animal with low mobility, terrestrial gastropods are slow movers which might reduce their capability to recolonize burned areas. Santos et al. (2009) studied this group of animals at four years after fire in a very large fire (>4000ha). There were few changes in the number of individuals but more in species. Burned shells indicated that many individuals were killed by fire. They found that differences inside the burned area or close to the edge were similar with the unburned forest, suggesting that refugia were a source of colonization. They argue that the larger the fire and the greater the fire-intensity would probably reduce the colonization capacity of these organisms. Unlike what it was found in southern France (Kiss and Magnin 2006), resilience was reduced and four years after fire community had not recovered to prefire levels.

Determining the origin of the colonization (the refugia hypothesis of the immigration hypothesis) is also important for micro-mammals and in the case of large fires. Studies (Prodon et al. 1987, Fons et al. 1993) about the dynamics of small mammals after large fires found that the sex ratio of the wood-mouse (*Apodemus syltvaticus*) is strongly unbalanced towards the males in the burned sites, which suggest that colonization from nearby areas took place due to the fact that males are more active disperses than females. The reduction in internal parasites through the years, in comparison with the initial ones, indicated that the first migrants carried parasites from their area of origin, which was outside the burned area, but later infections decreased due to the reduction of parasites in the burned area due to fire (Feliu et al. 1993). Consequently, fire size can affect colonization patterns of certain groups of animals and other interactions.

5.2.3. Fire intensity and severity

Two studies illustrate how changes in fire intensity and ensuing fire or burn severity can affect faunal communities and populations. A study on the recolonization dynamics after fire of soil micro-arthropod in a *Pinus halepenis* stand found that total number of micro-arthropods was reduced by nearly one order magnitude following fire (Henig-Sever et al. 2001). A similar reduction was obtained in Collembola with an even more drastic effect on the number of groups represented in the high fire-intensity areas. Fire directly or indirectly (increased ash content and pH) was a major factor in shaping the postfire soil invertebrate community.

A second example comes from birds, a group of organisms that because of their larger mobility would be expected to produce lesser changes in relation to fire intensity. Legume-shrublands (*Citysus purgans*) of the upper Mediterranean mountains of the Pyrenees are burned by shepherds to reduce encroaching and increase pasture availability. it has been found (Pons and Clavero 2010) that faunal values decreased as fire intensity increased. High intensity fires burned most branches reducing the capacity of shrub-dwelling birds to use the plants hence this group disappeared in the first year after fire and was forced to forage and nest in nearby unburned areas. By contrary, mild fires left dense burnt cover and unburned patches, favouring their presence.

5.2.4. Fire season

The effects of the fire-season can be due to the changes in fire intensity linked to different moisture content of fuel through the seasons, or to the state of the populations at the time of fire. For instance, early in the season fires are usually less intense, and therefore their direct impacts are likely to be less detrimental on the faunal population. The mortality of the tortoise *Testudo hermnnanni* in Greece was very different depending on the season of fire, from up to 47% in summer to <6% in winter (Hailey 2000).

For a given fire intensity the size and age structure of the population can have different impacts on it. Early season fires may affect faunal population in the reproductive phase, whereas late season could occur after dispersal. Consequently, the effects could be very important. An extreme example is the case of migrant animals. Depending on the date of emigration and fire, fires can occur when they have already reproduced and abandoned their sites, so that when they return the following year the vegetation will have recovered and, eventually, will be even more favourable for the species to reproduce. If the fire takes place before arrival from migration, the conditions will be most severe during the first summer, when vegetation will have not recovered. Since many migrants have strong philopatricity that means that the level of stress during that year will be very great.

5.3 Organism interactions in relation to fire

While the impacts of fire on plants and animals have been addressed normally separately and for particular groups, a number of interactions within them and across groups necessarily occur. To have a full understanding of the implications of fire for biodiversity an understanding of such interactions is needed. Yet, few studies have undertaken this. In a context of changes in climate and other environmental stressors is important to recognize them. Following are some examples that illustrate some of these interactions, which can cascade across trophic levels or interacting groups.

5.3.1. Plant-plant interactions

Changes in plant population and species can occur after fire due to competitive interactions among plants. When this involves tree species the effects can be significant. Seedling establishment of trees can be compromised due to the high competitive pressure from herbs, shrubs or other trees that thrived in a suppressed state in the understory but that are competitively released once they resprout after fire (De las Heras et al. 2002). Consequently, the probability that a given stand will be succeeded by the same type vegetation is not guaranteed (Retana et al. 2002).

5.3.2 Plant animal interactions

5.3.2.1 Seed predation

Seed predation in Mediterranean environments can be very large. In the absence of fire seed predation in *Pinus halepensis* has been reported to be up to 97% (Ne'eman et al. 2004). Granivorous rodents and, to a lesser extent, ants are responsible for it. After fire, large quantities of seeds are released from serotinous cones of pines (Daskalakou and Thanos 1996) at a time when granivorous rodent pressure is reduced due to attritions by fire. Yet, seed predation can be very high during the first year although it is not sufficient to jeopardize seedling establishment (Broncano and Retana 2004). Establishment depends on seed availability and conditions after fire, particularly rainfall patterns (Daskalakou and Thanos 2004). Consequently, large variations in seedling recruitment have been observed. When establishment during the first year is not insured, regeneration will rest on seed sources from nearby areas. However, after this time seed predation will be much higher due to the rapid recovery of rodent populations (Broncano and Retana 2004). Consequently, regeneration of certain key species may depend on the combination of certain biotic and abiotic factors.

5.3.2.2 Herbivory

Resprouting plants after fire have higher water content and are more nutritious, and herbivores may be attracted to burned areas. The capacity of Mediterranean plant to support herbivory is very large, and although browsing intensity and time may reduce growth depending on species (Espelta et al. 2006), no examples are available indicating an impact on plant mortality. Herbivory can be particularly important in seedlings, since attritions on these will have long lasting effects, and not all species are capable of resprouting when clipped (Verdaguer et al. 2001).

Of particular interest are the interactions of herbivores that could result in forest die back, hence in major alterations of forest fuels. Examples from some forest pests are illustrative. The pine processionary caterpillar (*Thaumatopoea pityocampa*) attacks various species of pines. High elevation areas are safe sites due to the cold that impedes the propagation of this pest. The cause of the cycles of this pest is unclear, but both biological and environmental factors may play a role. Warming trends are expanding the species upwards and in extreme years it has been detected to increase in elevation (Battisti et al. 2006). The interactions between pest and pathogens on plants can have important consequences in fire-prone environments. Increased pest pressure can reduce growth, seed production, establishment rates and mortality of branches or adult individuals (Hodar et al. 2003). Increased fuel loads can have important consequences for the risk of fire, particularly when it affects species that have no capacity to respond to fire, as it happens with *Pinus sylvestris*. Pests in combination with other climate stresses can significantly produce forest die-back (Allen et al. 2009) and be a source of altered fuel quantity and quality and of fire risk.

5.3.2.3 Pollination

Plant pollinators typically increase shortly after fire until the first few years, and asymptotically decreasing as the stand matures. This pattern follows floral abundance and diversity (Potts et al. 2003). Increased pollinators in the early stages after fire may, however, not imply increased pollinators services. Changes in pollinators diversity linked to shifts in the type of flowers dominating in post fire environments can cause reduce seed set in recently burned areas in comparison with unburned ones (Ne'eman et al. 2000). This is most surprising since nutrient content in burned areas should favour plant growth and, eventually, seed set in such areas.

5.3.2.4 Dispersal

In homogenising landscapes due to fire and abandonment, the probability of an area to be colonized by tree species depends, among other things, on dispersal capacity. Some birds (jays) are efficient dispersers of acorns and can transport them to long distances. Dispersal by jays to shrublands is much lower than to recently abandoned fields or forest (Pons and Pausas 2007). Since shrublands are the dominant type of vegetation after frequent fires it indicates that processes towards more mature stages may be reduced by changes in dispersal agents. Within burned areas little is also known about changes in dispersal mediated by animals. After fire, seed rearrangement mediated by mutualists can expand the range of species, as shown by (Arnan et al. 2009) who found that a chamaephyte with limited dispersal capability colonized new areas after fire due to ant dispersal.

5.3.3 Animal-animal interactions

5.3.3.1 Predation

After fire various guilds of organisms suffer dynamics that are controlled by external factors but also by interactions among them, predation being one of these factors. Understanding the changes along succession in predator-prey interactions would allow better comprehending possible mismatches. A number of examples indicate that in recently burned environments prey abundance may exceed predators presence (Herrando et al. 2005). The study found that there is relationship between preys (arthropods in this case) and insectivorous birds along age-since burning. However, in the early stages after fire there was relatively more prey than birds. Consequently, prey abundance did not limit bird presence. (Torre and Diaz 2004) found as well that the abundance of small mammals was larger than expected from vegetation structures in recently burned stands, whereas the contrary occurred in long-unburned ones suggesting that predator pressure, low in the early stages after fire, and higher in the more mature phases, might be responsible for this pattern.

5.4 Postfire management effects

5.4.1 Effects on plants

In burned shrublands and grasslands postfire management actions are usually reduced, except in the cases of limited vegetative cover and high risk of erosion. Postfire management actions usually aim at incrementing vegetative cover by seeding. The effects of these have not been studied but limited effects are expected. More important is when forest are burned and logs are extracted for timber of other purposes. Experiments aimed at evaluating the effects of salvage logging on plants indicate that these actions may not be important (Pérez and Moreno 1998, Bautista et al. 2004). However, negative effects on the ecosystem are foreseen in certain soils when erosion gullies can form (Bautista et al. 2004).

5.4.2 Effects on animals

Studies on the impact of removing the logs or cutting and letting them down onto the soils have yielded contradictory results for animals. Removing the trees significantly impact the bird community in a forest in Eastern Spain (Llimona et al. 1993). Burned logs were an important resource for various species and for different functions. In a *Pinus halepensis* forest of Israel, (Izhaki 1993) compared the bird dynamics in plots that had been left intact, cut and laid down or removed. He found that cutting significantly increased bird richness. By contrary, in the same area, (Haim 1993) found that cutting was more detrimental to the rodent community. This suggests that open space favour the colonization of new species, and cutting the logs may favour this. Since some of the species of open spaces are more important from a conservation point of view than those persisting from the forest (Pons 2003), which are usually more widespread in Europe, it implies that postfire management can be used to favour conservation purposes. Nevertheless, seed dispersal aided by forest birds might also be of value, as well as the regulation of insect populations. Consequently, each management action after fire must carefully consider its objectives since the effects on different groups might be diffferent.

5.5 Landscape interactions

5.5.1 Plants

Plants tend to regenerate after fire mainly from endogenous sources (by resprouting of from on-site store seeds). Consequently, the interactions between changes in landscape patterns and plant species richness has not been much studied. Nevertheless, the relative distribution of species of each group can vary depending on environmental characteristics and these could change due to fire. The number of seeders species as well as the seeder/resprouter ratio increases towards Mediterranean conditions in comparison with more moist habitats (Lloret et al. 2005). By contrary, fire sensitive species increases towards moisture areas. Fire changed these proportions in a way that the seeder and seeder/resprouter ratio increased in areas with greater fire activity and so it did with the fire sensitive proportion of species. Resprouters were similarly represented across the climatic gradient. This implies that more seeders are likely to occur in more areas as fire expands and that the regeneration response will be more dependent on postfire climatology. Therefore, large scale differences in species pools might occur as a result of fire.

The filtering effect of fire on species pools may not be restricted to differences in regeneration modes. Moreno (unp. data) found that in a very large fire (18000 ha) three years after fire species richness increased in the burned area with respect to the unburned surrougindgs. However, increases were not equal for plants differing in the distribution range and differed also by substrate. Cosmopolitan species increased in both substrates, and their increment was much higher than that of the other groups. Species with a reduced range (Iberian Peninsula endemics) increased very little and differently depending on the substrate (Fig. 8).

This suggests that landscapes in interaction with fire may play a role in altering species abundance at different scales. It appears that cosmopolitan species can take greater advantage of fire than species with a reduced range. The reasons for that change are not known. Different plant functional groups might be differentially sensitive to heat or respond differently to the changes in the environment induced by fire, in particular during the critical phase of post fire germination. Luna et al. (200) showed for species of two life-forms (chamaephytes and hemicryptophytes) that host the largest fraction of endemics (Melendo et al. 2003) that heat tolerance among plants that differ in range size was different, whereby Iberian endemics and resprouters were more sensitive to heat that non-sprouters and widely distributed. Additionally, light exposure, which increases in fire environments, affected negatively to Iberian endemics and chamaephytes (Luna and Moreno 2009). Furthermore, sprouters and endemics to the Iberian Peninsula were more negatively affected by cold exposure, than non sprouters and widespread species (Luna et al. 2008). This means that if they miss the first rains in autumn germination would be delayed, with possible risks due to predation.

Even more, at this same site, the relationship between species rarity (range-size, local abundance) and germination-niche breath differed among chamaephytes or hemicryptophes and it was such that germination niche-breadth was positively correlated with range-size for chamaephytes, but negatively for hemicryptophytes (Luna and Moreno 2010). This means that species that differ in their rarity might be differentially sensitive in their germination requirements, which could possibly influence their sensitivity to fires. Much more needs to be known before we can make predictions as to which species are likely to suffer most from ongoing changes in fire regime. Nevertheless, it appears that the traits of some endemics makes them candidates to be at greater risk due to increased fire frequency, and that patterns of abundance at landscape levels might be affected by it.

5.5.2 Animals

Among animals, some have greater mobility than others. It is expected that those with greater mobility (e.g., birds) might colonize burned areas more easily and that the colonization process should be driven by niche requirements more than by dispersal limitations. Therefore, colonization patterns by this group should be rather homogeneous. However, authors (Brotons et al. 2005) have found that various recently burned areas differed in the species composition of birds and that colonization was related to sources of dispersal from nearby areas. Consequently, dispersal processes determined colonization rates and landscape heterogeneity can play an important role in determining colonizers after fire.

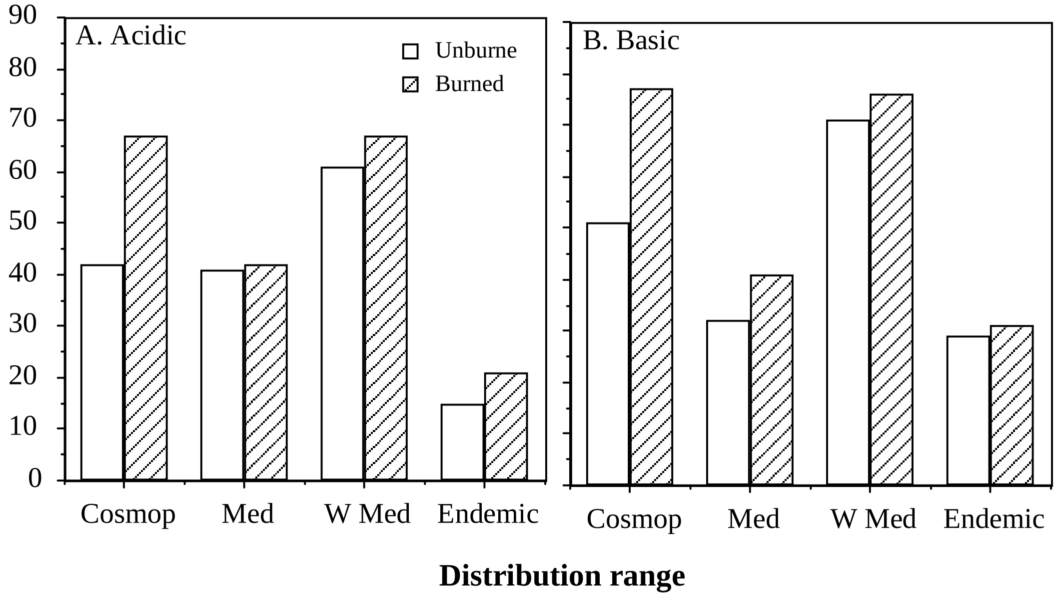


Fig. 8. Number of species of various distribution ranges in unburned and burned *P. pinaster* (acidic) and *Pinus halepensis* (basic) stands around and within (respectively) a large (18000 ha) fire in Eastern Spain. Cosmop: cosmopolitan; Med: Mediterranean; W Med: western Mediterranean; Endemic: Endemics to the Iberian Peninsula. From Moreno, Luna and Pérez (in prep.).

Taking into consideration that faunal (bird) diversity decreases in scrublands in comparison with agricultural or taller shrub or forest areas, but that scrublands can also contribute with additional species not common in any of the previous ones (Moreira et al. 2001), a question of interest is to know how fire occurrence and landscape changes affect faunal biodiversity at large scale (Gil-Tena et al. 2009). This study found that, at a large scale, bird diversity was not negatively affected by the combination of forest expansion and increased fire occurrence. Actually, the expansion of an endangered species seems to have been driven by fire occurrence (Brotons et al. 2008). It appears that in the Mediterranean wildfire impacts will not necessarily be always negative for vertebrates at certain scales depending on the fire regime and rate of landscape change. A fire regime characteristic of smaller fire sizes could even be advantageous for the maintenance of species diversity of vertebrates, not so one of large fires (Moreira and Russo 2007).

6. FUTURE TRENDS

6.1 Land-use and land-cover

Projections for the last part of the century indicate that the process of abandonment of marginal areas or its utilization for other uses will continue, with further concentration of agriculture in more fertile areas. Additionally, EU and national policies are promoting reforestation with native species in marginal areas, to favor carbon sequestration. The surface devoted to agriculture is projected to decrease depending on the emission scenario considered. For instance, less emission-intensive scenarios (such as the IPCC SRES B1) produce reductions in agricultural areas for Spain of 33% by the year 2080. On the other hand, more emission-intensive scenarios (A1) project reductions of up to 70% for the same date (Rounsevell et al. 2006). Consequently, it is foreseeable that the process of land-use change will continue during this century. A reduction of the surface devoted to agriculture could imply an increase in the surplus land dedicated to other uses, mainly forests in all its facets, including wildland areas, with the corresponding increment in fuels and hazard for these landscapes.

6.2 Climate

Annual temperatures are projected to increase in southern Europe and the Mediterranean (SEM) more than the global average (for the A1B scenario and the period 2080-2099, the projected global warming is 2.8ºC, whereas for the SEM is 3.5ºC). Warming will be largest towards the south and the interior, and will be highest in summer (4.1ºC, for the same scenario and period). Maximum temperatures are likely to increase more than average or minimum temperatures. Warming will be greater with increased GHG emissions (Christensen and Hewitson 2007). Annual precipitation is very likely to decrease in most of SEM, and the number of wet days is very likely to decrease. Precipitation changes will not be homogeneously distributed between seasons, with summer precipitation tending to experience the greater reduction (24% reduction in summer vs. 12% reduction in the annual total for the same period and scenarios as above). Precipitation changes will vary throughout the region, and the greater reductions are likely to occur more towards the south. Other changes for SEM include decreases in relative air humidity and cloud cover, particularly in summer; no significant changes in 10-m mean annual wind speed are expected, except for a light increase in summer (Christensen and Hewitson 2007).

There is uncertainty with regard to extreme, summer, short-term precipitation in the Mediterranean. Much larger changes are expected in the frequency of precipitation extremes; for SEM, large increases in the frequency of low summer precipitation are projected. The number of dry spells and the risk of drought is likely to increase in SEM (100-yr droughts are expected to occur under certain scenarios by the end of the century every 10-yr or less), notably in southern Europe (Lehner et al. 2006).

6.3 Biodiversity

During the last century, increases in global temperature have affected ecosystems and species throughout the world. Observed changes affect the genetic of populations, their phenology, physiology and distribution, among other (Parmesan and Yohe 2003, Root et al. 2003). In general, advancement of phenological phenomena towards spring and delays in fall, and shifts in distribution range towards the poles and towards higher elevations are some of the salient features of the ongoing changes. Many of these changes can be attributed to the ongoing anthropogenic climate change (Rosenzweig et al. 2008). Climate change can commit to extinction a sizeable number of species in the world (Thomas et al. 2004). Following are some details of the expected impacts of climate change in European plants and animal groups.

6.3.1. Plants

Niche-based modelling for a large set of European plants show that species losses from certain areas by the end of this century could be very significant (+40%)((Schroter et al. 2005, Thuiller et al. 2005). Patters of estimated species losses and turnover do not vary very much for different emission scenarios. The Northern Mediterranean (52%), Lusitanian (60), and Mediterranean mountains (62%) environmental zones are the ones facing the greatest losses, with the Boreal (29%), Northern Alpine (25%) and Atlantic (31%) being the less sensitive. Overall, for all species studied it was calculated that half of the species studied could become vulnerable or endangered by 2080. These results are consistent with earlier studies (Bakkenes et al. 2002). Incorporating in the models other interacting factors, such the effects of CO2 on water availability, significantly altered the results, different groups of species being differentially sensitive to this (Rickebusch et al. 2008). Species change and functional diversity change will vary across Europe (Thuiller et al. 2006). While the results emerging from these studies cannot be taken as forecast due to the inherent limitations in the modelling approaches, they indicate that the potential for plant species change is very great.

6.3.2. Animals

Climate change-based projections for all European, non-volant mammals based on two emission scenarios indicate that, assuming no migration, between 5-9% of the species risk extinction, while 46-78% may be severely threatened with losses greater than 30% of their current climate-space based distribution. Results are less severe, but also significant, even assuming unlimited migration capacity. Range losses are also likely to lead to reductions on population size, with its negative effects on population viability. Species richness is dramatically expected to decrease. Among the most vulnerable species are those in the South and in Islands. Species richness is likely to increase in the Northeast and at higher elevations (Levinsky et al. 2007).

Modelling European breeding bird distribution by the end of this century based on several GCMs projects major shifts in avian distribution (Huntley et al. 2006). The potential distribution range will be reduced to 81% of what it is today, the majority of species having 50-80% distribution of their current one. Furthermore, the overlap between future distribution and that of today is, on average, 39%, with a large number of species having no overlap at all. As a consequence of the displacements the areas of highest avian species richness shifts northeast, with areas in the south and east losing the greatest number. This results in an average reduction in species richness of 8.6% for Europe.

Projections for European amphibian and reptile fauna by the middle of this century indicate that the majority of them could expand their range based on various scenarios and GCMs under the assumption of unlimited dispersal. However, if they were unable to disperse most species (>97%) would reduce their range. Loss of suitable climate space is projected to occur mainly in the southwest of Europe, including the Iberian Peninsula. This is due to the fact that dry conditions in the south west are projected to increase, approaching the levels found in North Africa, where few amphibian species are able to persist. Species in the south-east and, particularly, in the Northeast are projected to gain suitable space. The projected reductions in rainfall and water availability, as well as the reduced dispersal capacity of many of these organisms making them rather vulnerable to climate change (Araújo et al. 2006).

These three examples illustrate well that the potential for climate change to induce species change is very great in Europe. There are numerous examples that the process of change is well underway. While the exact outcomes are difficult to project, different groups will react differently to accommodate to the new climate, and there are indications that even though organisms are reacting, shifts in distribution to accommodate to the new conditions lag behind those of climate (Menendez et al. 2006, Devictor et al. 2008). That means that many organisms are already living under stress since the areas they populate do not match their requirements. Differential responses to changes in temperature is causing mismatches in the interactions among organisms (Robinet and Roques, Gordo and Sanz 2005).

6.4 Trends in fire

6.4.1 Fire risk

Increased temperatures and reduced precipitation, altered precipitation patters, whereby rainfall concentrates towards the winter, will very likely cause increases in fire danger conditions in current areas in the SEM and extend these to areas in Central and, particularly, Eastern Europe in which fires were now not frequent or absent (Moriondo et al. 2006). The fire season will be longer, in some places lasting almost the whole year, and more severe, that is greater number of days of risk, greater number of days of extreme risk (Fig. 9). Increased dry spells and droughts (Lehner et al. 2006), and higher temperatures, particularly maximum temperatures will very likely increase the frequency of extreme fire danger conditions and with it the probability of fire, particularly of large fires (Moreno 2005, Moriondo et al. 2006, Giannakopoulos et al. 2009, Moreno et al. 2010).

The more severe conditions will very likely conduce to greater fire activity in SEM. (Carvalho et al. 2010) projected past fire occurrences into the future under climate change scenarios which resulted in very significant increments in the number of fires (279%) or area burned (478%). Modelling projections for Spain (Schroter et al. 2005) yielded increments of 55% to 112% for different emission scenarios. Since most fires are caused by people, extrapolating past fire occurrence into the future based on biophysical changes must be made with great precaution. Past records show that similar fire weather conditions can render very different fire occurrences. For instance, while the heat wave of 2003 produced the largest ever fire episode in Portugal, the adjacent territories of Spain, which suffered similar conditions, did not result in an extreme fire situation. However, the few fires that were ignited during those conditions resulted in large fires. This is consistent with the finding that fire-size inequalities tend to be greater in more severe years and among higher fire severity areas (Vázquez and Moreno 1995).

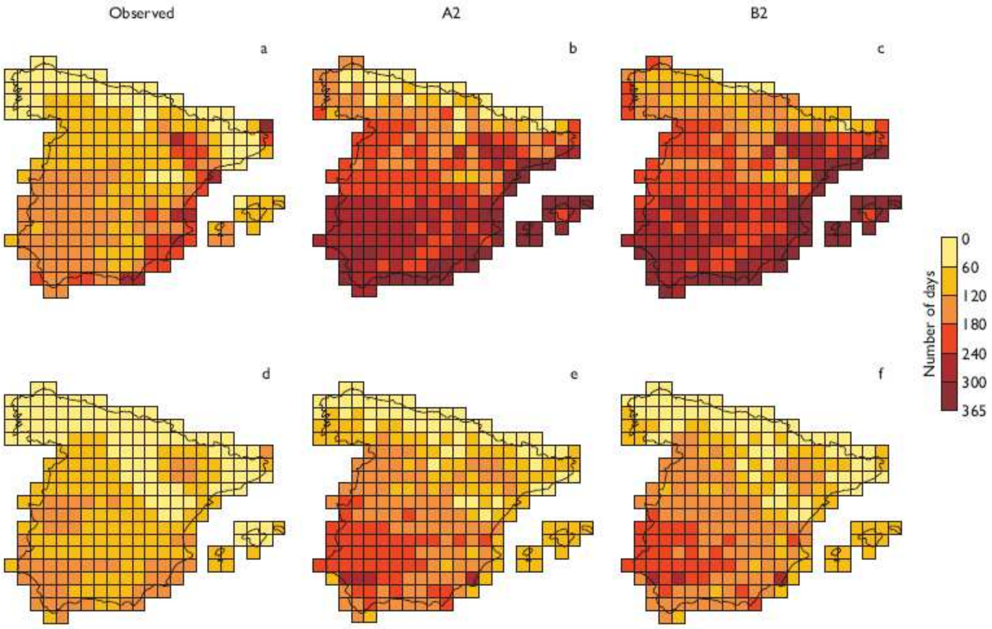


Fig. 9. Observed (a, d) and modelled (b, c, e, f) values for the Period of Alert (a, b, c)(number of days between the first and last day during the year that FWI≥15 continuously for a week) and Period of Risk (d, e, f)(number of effective days during the PA in which FWI≥15) during a 30 year period in Spain. Observed values are based on daily data of the MARSSTAT database from the Joint Research Centre of the EC at Ispra (IT), and the period 1975-2004. Modelled data are the median of the A2 and B2 SRES scenarios of 5 Regional Climate Models with daily data for the period 2071-2100, made available by the Spanish Institute of Meteorology (Madrid, Spain). Cell size is 50x50 km. From (Moreno et al. 2010).

In the boreal region of Europe, simulations show a decrease in fire danger in the North and an increase in the South for a doubling of CO2 scenario (Flannigan et al. 1998). Calculations for Finland and the A2 scenario show that fire risk increases from the North towards the South. In the south, the number of days of fire alarm can increase from a maximum of 100 in the present to 160 days by the end of the century. In the north these figures increase more moderately, from 30 to 36 days. The expected increase in the number of fires in the country is about 20%, the greatest frequency being in the South with an increase of 30%, where 9 fires per year and 1000 Km^2 are projected (Kilpelainen et al. 2010).

In summary, with climate change there will be an increase in forest fire potential that is greatest at SEM, most particularly in the Iberian Peninsula. Areas of Eastern Europe, where fires have been uncommon until now, might also be affected. Among the most relevant projections are those concerning the maximum number of days of extreme risk. Unfortunately, current models cannot simulate well these situations, so much awaits to be learned yet. The episodes of heat waves, drought and widespread fires throughout large extensions across various countries, of which 1994 (Spain), 2003 (Portugal) and 2005 (Greece and the Balkans) are worth mentioning as to what future fire risk could be like.

6.5 Other interacting factors

The interaction among drought and regeneration is a particular issue in the MR under future climate since the postfire regeneration potential of many areas will very likely suffer from reduced precipitation and increased probability of drought. If, as it happened in the past, drought is associated with large fires and widespread occurrence, this means that the potential for recovering after fire of large extensions might be in peril, thus adding capacity to vegetation change. Seedling recruitment under simulated drought is reduced, and is not related to patterns of mortality of adults, which implies that droughts and fire can result in significant vegetation shifts (Lloret et al. 2009). Synergies between drought and fire can significantly alter vegetation composition even under low warming scenarios (Fyllas and Troumbis 2009). Increased drought may shift fire-prone areas towards more northern areas in Europe, increasing tree mortality. However, along the transition between different vegetation types, large amounts of decaying fuels may be exposed to increased meteorological fire danger conditions giving way to greater fire risk (Peñuelas et al. 2010).

Climate change is likely to reduce nutrient availability and turnover (Sardans and Penuelas 2007), reduce soil moisture and, ultimately, reduce growth and primary productivity in semiarid areas, particularly as they become drier. However, in many upland areas, where productivity is more controlled by temperature than humidity, increased temperature could increase productivity in a first period (Gracia et al. 2005). This would add greater capacity to the vegetation to quickly recover after fire and sustain additional fires provided ignition sources. Something similar could occur along latitudinal gradients as southern biomes shift northwards (Metzger et al. 2008). How fire would affect current, non-fire prone areas is still poorly known. However, a study (Thonicke and Cramer 2006) found that the modelled impact of fire on the vegetation of southern Germany would not be important.

Postfire environments are rich in resources and competition in their initial stages is reduced. Consequently, disturbances can open space for the colonization of exotics (Gritti et al. 2006). These authors modelled the potent for invasions of Mediterranean areas in relation to climate change and disturbances. They found that invasibility varies between dry and moist areas and in relation to disturbances. They concluded that Mediterranean island ecosystems are likely to be dominated by exotics. Fires, particularly large fires, can open spaces that could be potentially used by high-dispersal invaders. Invasives, in turn, can modify fire regime and further increase invasive potential (Vila et al. 2001).

7. THE CHALLENGE OF MANAGING AND CONSERVING BIODIVERSITY IN FIRE PRONE AREAS OF EUROPE

7.1 Landscapes, climate change and fire interactions

Much of southern Europe can be characterized as having landscapes highly sensitive to a number of drivers that interact with fire and in which fire interact with them (Fig. 10).

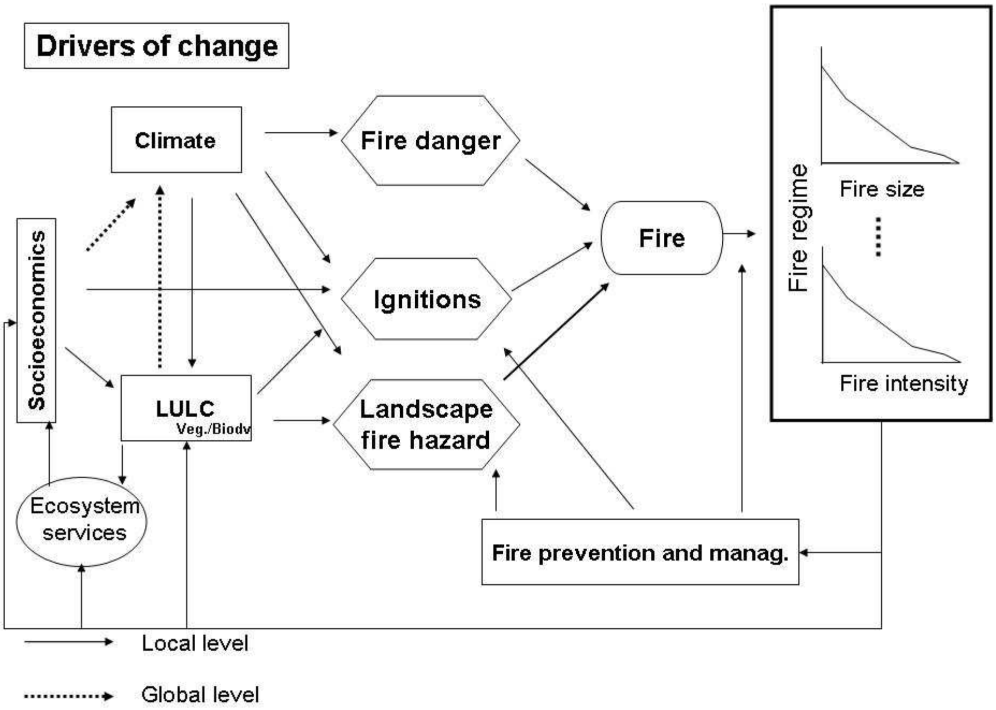


Fig. 10. Diagram showing the complex interactions that govern fire. Socioeconomic drivers affect the global economy and, through it, the global and local climate and land use/land cover (LULC). These determine fire danger, ignition sources and fire hazard. Fire spread will depend on fire fighting and other management measures. Fire regime is the result of these interacting factors. Fires can modify the landscape, including biodiversity, and affect future fires as well as ecosystem services. Changes in drivers will affect fire regime.

Socioeconomics interacting with the physical and biological background determined the use of the land. Changes in this and in climate determine changes in the landscape and, directly or indirectly, affect ignition sources and fire occurrence and regime. Fires feedback on the landscapes and reconfigure them, making them more homogeneous and prone to further fire. Climate change will modify these interactions and, ultimately, fire regime. Despite the semiarid conditions reigning in much of SEM, the system is still productive enough to quickly grow vegetation that can feed forward on fire. This, however, might change under future climate, and the times for recovery after fire might be delayed, which means a higher probability that fires will affect systems in recovery. Socioeconomics, by changing modes of life, and providing ignition sources or increasing the capacity to deter fires, could affect this positively or negatively if ignition sources are contained. The negative trends in area burned in the last decades in some countries indicate that this is possible. Nevertheless, under more severe climate extreme episodes of fire will with most certainty occur.

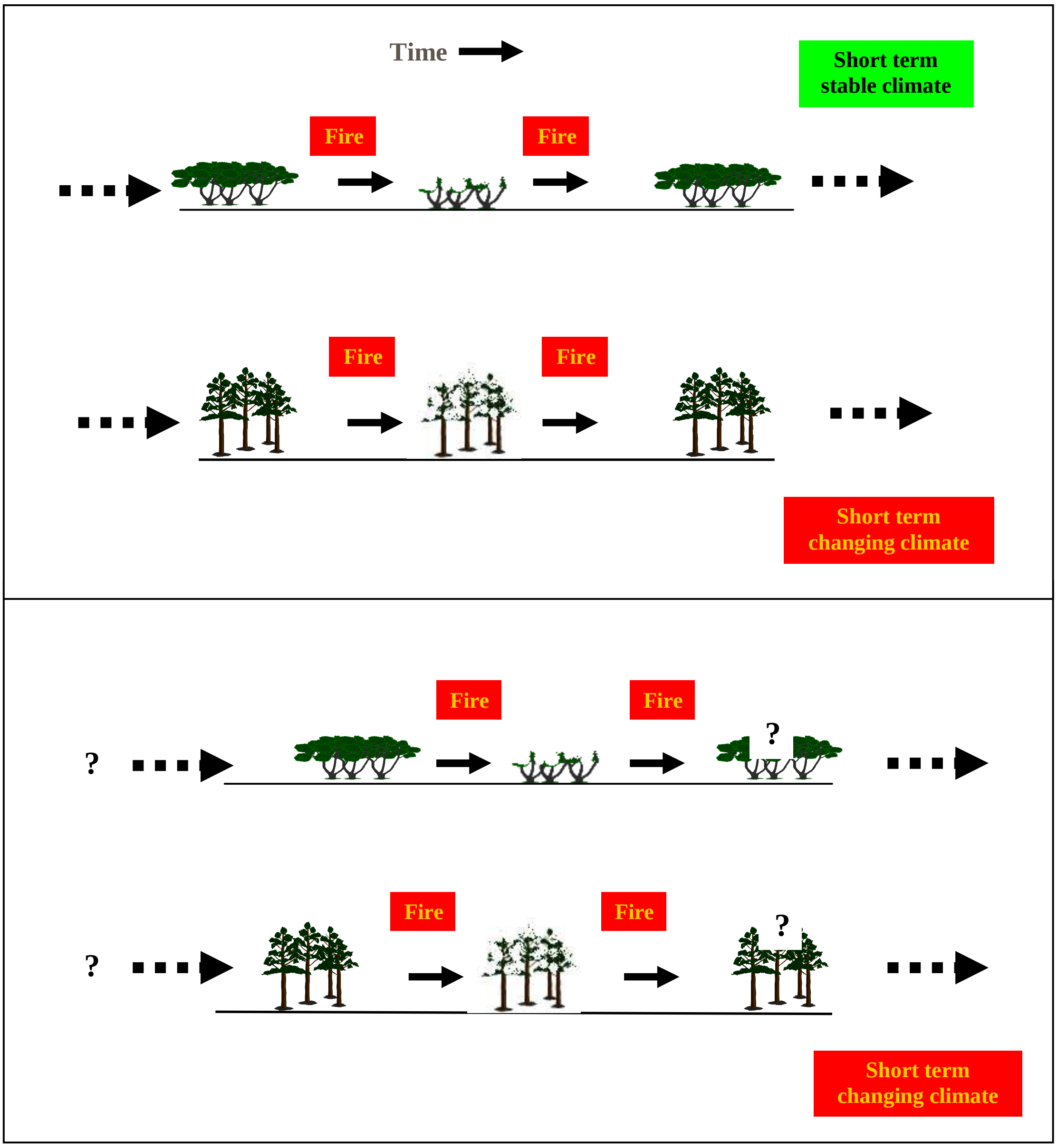


Fig. 11. Scheme of the classical fire paradigm (top panel): Vegetation at a certain point in time is traceable to its fire history. Fire sets the system into a regeneration state that, provided a short term stable climate, some time later will result in similar structure and composition. Under the new situation, changes in climate would result in uncertainties. By contrary, in highly humanized systems, as in the Mediterranean Region (bottom panel), the initial stage is not traceable to fire history but to a combination of actions by man that may include fire. After fire the system may not necessarily return to its prefire state even under stable climate. With a changing climate the uncertainties are much greater.

7.2 A system in transition

In regions of the world that have not been so severely altered by man, the fire-vegetation paradigm can be characterized as follows: vegetation at a given place is the result of past fire regime; different places differ in their successional stage after fire, and these follow their course until a new fire starts the process again. In this case, the environment imposes their constraints and variability should be attributable to this and disturbances as fire. Natural climate change and variability would be the factors that would modify the system response at the time scales they operate (Fig. 11).

However, in areas with such a long history of human occupation such as in SEM, the vegetation at any given place is the result of the multiple interacting factors associated to the various human activities. The differences among patches in a landscape are related to the environment but mediated by the role of man. The origin and regime that characterizes any piece of vegetation is probably unknown and was unstable through the years, as much as it was its land-use history. Extant vegetation and species assemblages may be in transition after cessation of traditional use of the land.

Parallel to vegetation changes, landscape changes may also be shifting from a coarse grain pattern, more linked to the traditional use of the land to a coarser grain one as fires take over the control over the landscape (Fig. 12). Patterns through time can change depending on changes in fire risks associated to climate change, among other.

Difficult as it is to project future impacts of climate and other global changes on the vegetation and species composition of any system in the first type, much more difficult it is to do so in SEM areas. Restoration has no easy models to use them as a reference, and many ideas need to be revisited at the light of new paleo-ecological evidence. Given the threats of changes in fire and other climate and global changes over the values at hand, not the least its distinct and rich biodiversity, the challenge of conserving these territories under the ongoing climate and land-use/land cover changes and other global changes is paramount (Fischlin and Midgley 2007).

8. ADAPTING TO CHANGE: RECOMMENDATIONS FOR CONSERVING BIODIVERSITY UNDER FUTURE CLIMATE CHANGE AND WITH FIRE

8.1 The role of fire must be included in conservation of species and habitats in fire prone areas

Fires have been occurring, and will most certainly occur within many protected areas in southern Europe and in the landscape matrix that surrounds them. Fires are generally considered as a threat, and fire suppression is the dominant policy throughout SEM. There are enormous skills and capacities to fight fires. Yet, when they break out inside or around protected areas they will burn through them. But since the main/only policy is to fight them, provisions to understand how they directly or indirectly affect protected areas and species once burned are, for the most part, lacking. Until now, the ecological role of fire is ignored. Consequently, when they occur there is no contingency plan as to how the affected system will be impacted. Therefore, even without any climate change, biodiversity conservation plans need to consider how fires will affect species and habitats throughout the territory. Fire ecology is a must in all management and conservation plans, and strategies to incorporate this knowledge must be enacted.

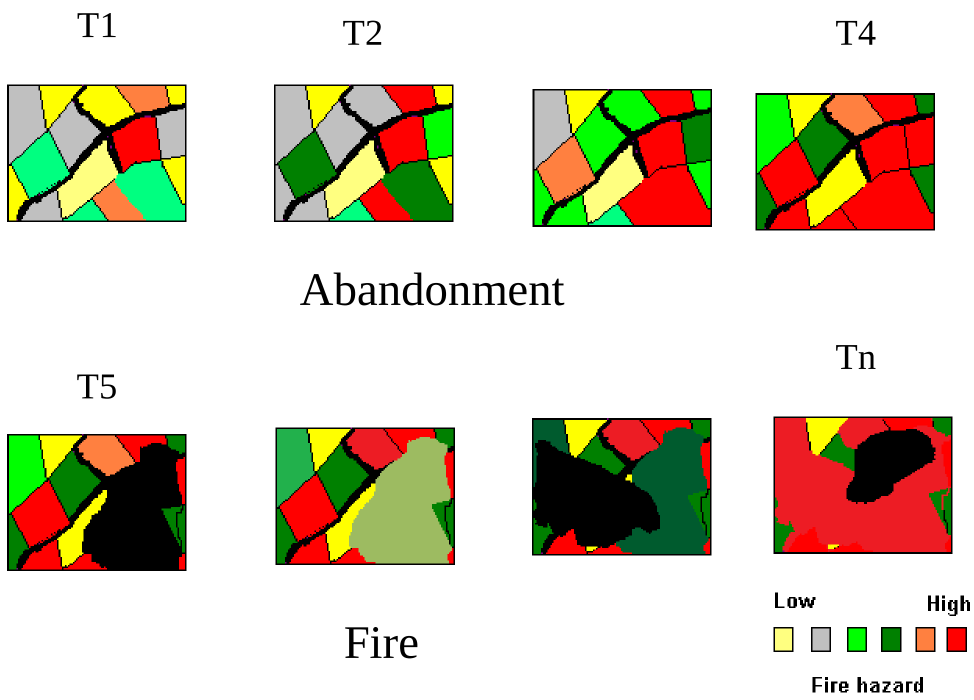


Fig. 12. Schematic view of changes in an imaginary landscape after cessation of the traditional use of the land, followed by abandonment and fire. The fine-grain mosaic landscape turns into a coarser grain one as abandonment proceeds through time, and more so as fire takes over the control of the landscape. Changes in fire regime will further modify these patterns.

8.2. Fire can play a role for fire in conservation

Some ecosystems and species depend on fire or can benefit from it. Identifying them might be critical since current policies will jeopardize their persistence. In these cases, plans for introducing fire, either by prescribed burning, or, when appropriate, with wild fires within acceptable conditions to avoid other risks must be made. Because the prevailing view is that fires are undesired, and the risks that entail managing fires is great, conservation plans in need of fire must be implemented with great care to avoid accidents that would stop the continuation of needed plans with the concourse of fire.

8.3. Conservation plans aimed at specific target species must consider how fire will affect them

Species or groups of species are impacted by fire differently, depending of fire characteristics and other factors. In the case of protected areas whose objective is one or a group of particular species, the viability of their conservation in a context of fire needs to be specifically considered. Management plans that address the possible impacts of fire need to be species or group specific, since different species are likely to respond differently to fire.

8.4 Vulnerability of the protected areas network to fire

Corridors and stepping stones are important elements for insuring population persistence and species migration, more so in view of the impending threats. These elements, however, may be subject to fire. When these components are formed by forest, fire can alter their functioning capacity for long. Since it is very likely that some of these more isolated elements are in areas with greater human influence, their susceptibility to fire and repeated fire might be rather great and needs to be quantified since its long-term persistence may be severely threatened. As with the rest of the protected areas, the impact of fire needs to be known in advance in order to better evaluate their capacity to continue playing their role. Robust network designs, capable of not succumbing to a single fire, are needed to allow these places continue playing their vital service.

8.5. Urban developments and roads near protected areas threatened their persistence

Most fires are lit by people. Towns and roads are the main sources of ignitions. However, the probability of burning is still high at some intermediate distance to roads and towns since fire can travel long distances. Protected areas within these domains are at higher risk of fire than those further away. Urban developments into the wildlands and near protected areas can be a threat to these due to increase ignition probability and subsequent fire. Also, the network of roads crossing protected areas, in addition to other perils, can clearly add risk. These two elements must be cautiously considered when declaring protected spaces and be particularly monitored during the time of high fire risk. Eventually, specific restrictions might have to be put in place to minimize risks. Risk mapping of protected spaces taking into consideration proximity to roads and towns is critically needed.

8.6. Identifying synergies/conflicts between fire and conservation

Fire fighting includes, among other, fire break lines or fire-break areas. These can provide open space and hence favour species persistence different to those in the preserved matrix, particularly when these are forest. The role of such areas and corridors as sources of rapid colonization after fire needs to be appraised. These areas can serve as colonization points but there are positive or negative elements (increasing potential for invasive species) that need to be fully considered. The advantages and disadvantages of these areas in the event of fire need to be taken into consideration.

8.7. Changes in the landscape matrix through fire

Abandonment will continue in response to changes in socioeconomics and with climate change. Abandonment modifies the landscape matrix towards homogenization and that can threatened the persistence of many species. Fires can open up space and introduce large changes in the landscape matrix. Not all organisms will be equally affected but such changes in the landscape structure. Some, through the openings made by fire, will be favoured. Others will be negatively affected. Conservation plans must therefore contemplate the landscape scale changes that are introduced by fire.

8.8. Assessing future risks

Changes in fire frequency, intensity/severity, size and season must be specifically contemplated for conservation areas under scenarios of climate and land-use/land cover change. This must be done for current areas with fires and for those in which fires were not present but that are likely to occur due to the changes in climatic conditions and other factors. Each of the parameters that define the fire regime can differentially affect the various species. Changes in fire season, particularly when migrant species are concerned, need to be cautiously considered. Consequently, the impact of each of them needs to be assessed in general or for the particular species or group of species that are of interest.

8.9. Fire, species conservations and other stresses

Conservation scenarios that include fire must take into consideration the level of stress being endured by the various species since, little by little, they will inhabit areas that are more stressful for them due to changes in climate among other stressors. The capacity of particular species or groups to respond to fire under such circumstances and to changes in fire regime needs to be appraised. As fires might occur under extreme conditions not seen until now (drought being the most relevant) this type of interactions need to be fully taken into consideration in future management plans for biodiversity conservation. Additional stresses due to more frequent and intense heat waves, particularly in the open habitats of the first years of regeneration after fire, must also be known.

8.10. Conservation plans must include worst case scenarios

Although the great majority of fires are of small size, some of then can attain very large sizes, in the order of thousands of hectares. In Spain, the maximum size of any fire recorded is around 30.000 ha, and the maximum length is 45 km (Moreno et al. 1998). The potential for one fire to spread over a whole protected area at once is not negligible. Smaller and homogeneous areas in a matrix of high fire risk are the most threatened. The prospect of increasing fire size under future conditions further adds to this. Consequently, worst case scenarios that include burning a large portion or even the whole protected area when these do not exceed several thousands of hectares needs to be contemplated. The role of buffer zones in this context needs to be equally appraised.

8.11. Fires as opportunities to accommodate species to the new climate

Fires, by opening new space, and by having reduced competition among organisms in the early phases can open new space for species to move upwards or northwards in search of suitable climate. But this can also be used for invaders. Differentiating the new colonizers that are now attuned to the new conditions from those invading is important. Identifying the potential for fire to act as stepping stones must also be considered.

8.12. Identifying species at greater risk

Species of late successional stages, thus requiring longer time to colonize burned areas, are probably the ones at greater risk in scenarios of increased fire frequency. Moister sites should regenerate quicker than more xeric sites, but their rate of recovery will be delayed with the onset of reduced precipitations under future climate for large parts of SEM. Consequently, their recovery period will be extended and the probability of burning again in earlier stages of regeneration indicates that species proper of mature successional stages might suffer. Studies should emphasize determining which groups of species enter at which state of the postfire succession and on the time needed for their recovery.

8.13. Bottle-necks

Among those species most likely to suffer from fire are those of reduced distribution that are linked to particular systems that are fire sensitive. That is, those that do not regenerate after fire. A fire, particularly a large one, can severe these populations for long, making its recovery difficult. Identification of bottle-necks and deadly-traps among organisms and their systems in the event of fire is critical for those species that may be most threatened.

8.14. Species linked to others

Climate change is producing mismatches among species (in pollination, in dispersal). Furthermore, fire con contribute to alter them. Identifying mismatches that are enhanced by the combination of fire and climate change might be of relevance for the maintenance of species that may already be in danger.

8.15. Changes in the protected network

With climate change, the size of the protected areas will have to be increased to achieve the same conservation objectives. Until now, fire has not been taken into consideration in the design of the network of protected areas. Yet, its effectiveness can vary. Consequently, future modifications must consider how fire would affect its effectiveness. Since it is likely that the protected areas of the current network are those in a better state of conservation, which, presumably, are those further away from human influence, it is likely that new additions will be closer to human habitations, thence with higher risk of fire. Risk of fire must be included at the time of modifying the network of protected areas.

8.16. Educating on the ecological role of fire

Fire is commonly seen as something negative, but it can play a dual role in the conservation of biodiversity. Fire, for the most part, hardly receives any attention in education, even in university programs, or not as much as its relevance demands. Every effort must be done to form and inform the general public and students at all levels about the role of fire in ecosystems and biodiversity conservation.

8.17. Research needs

Knowledge on how fire affects the various groups of organisms across gradients is still a must. Long term observation sites, where the main groups of are studied jointly should be established.

Large fires, particularly large fires episodes, are laboratories that should be explored in depth for their role on biodiversity. Since many of these fires occur along gradients, these are opportunities that should not go by unexplored.

Maps with fire history are now possible for the last decades. These offer opportunities to study the impact of repeated fires on biodiversity across groups and across landscapes.

Protected areas are not static and will change with climate change. Modelling their fate and their vulnerability under scenarios of climate and fire change are crucial to understanding their future role in biodiversity conservation.

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