B2. Fire. Ramón Vallejo and Alejandro Valdecantos

1. The phenomenon of large wildfires in the world: causes and consequences

Large fires are natural in many regions of the world where vegetation fuels get flammable during the dry season or during dry years in wet regions. In the context of UNCCD, dry sub-humid to semiarid regions are especially prone to large wildfires. Drier areas do not support continuous vegetation; therefore fire does not sufficiently propagate so as to produce large fires. Human activities in dense populated areas are an important source of ignitions and a major driver of desertification processes. Human activities have increase fire frequency and changed fire regime in many regions of the world. According to the relationship between ecosystems adaptations and fire history, we may distinguish between fire adapted and fire sensitive ecosystems. In the former, fire is an essential ecological force that determines landscape shape, structure and diversity that require fire for regeneration. Fire sensitive ecosystems are those that do not commonly burn and do not show fire adaptations, and as a consequence are prone to suffer degradation caused by fire.

Figure 1. Forest fire impacts. The need to undertake mitigation, rehabilitation or restoration actions in burned lands derives from the identification of post-fire negative impacts.

In the Northern Mediterranean countries, the socio-economic transformation over the last fifty years from rural to urban societies, and the subsequent reduction of grazing, firewood exploitation, and abandonment of croplands has led to a dramatic increase in vegetation fuels as well as urban visitors of forests for recreation. The implementation of fire suppression policies has also contributed to the accumulation of high volumes of fuel, especially in forest understory, lading the fuel from the ground to the forest canopy. Afforestation practices in many Mediterranean countries were based on planting coniferous and eucalyptus with insufficient forest management in the established plantations. All these transformations led to an increase of fuel load and continuity and the spreading of fire-prone ecosystems on the landscape. A dramatic increase on large and high intensity wildfires has been one of the consequences (Fig. 2).

Figure 2. Evolution of yearly surface area burned and annual number of fires in the Region of Valencia since 1874 (top, Pausas 2004), and recent fire statistics in some Northern Mediterranean countries (period 1995-2004; bottom, data from FAO 2006 and Eurostat).

2. Forest fires as drivers of desertification.

2.1 Fire impacts on ecosystems and landscapes.

Fire directly affects vegetation, soil, and the less mobile fauna. Burned ecosystems miss totally or partially plant cover for a period of months to years. During this period, bare soil is exposed to wind and soil erosion, and topsoil degradation. At the landscape, land cover is modified, changing the interception, evapotranspiration and soil infiltration of rainfall, landscape structure usually becomes more homogeneous at large scales, and water, sediment and nutrient flow are altered, with frequent increase in runoff, hillslope soil redistribution, and sediment yield at the catchment scale. During the first one or two years after the fire, the risks of flooding and siltation are very much increased with respect to unburned catchments (Fig. 3), the same increase is experienced in relation to the offsite risk of damages to structures and population (i.e. by landslides or debris flows).

Figure 3. Non-regenerated pine forest catchment neighbouring unburned catchment in the Guadalest watershed, four years after fire (Alicante, Eastern Spain).

2.2 Effects on soils and water cycle

Figure 4. Key degradation processes occurring during and immediately after fire. See text for further explanation.

The impact of fires on soil properties and the subsequent consequences on soil erosion and runoff result from the combined effects of direct impact of heating during the combustion near the soil surface (severity), or through smouldering along dead roots, and the indirect effects derived from the loss of plant cover and forest floor after the fire (Fig. 4). The impacts of forest fires on the properties of soils depend largely on fire severity and fire recurrence. Severe fires, with high temperatures in the soil surface, produce loss of organic matter and nitrogen of the forest floor and topsoil, temporal soil sterilisation, soil surface crusting (especially on silty soils), water repellence modification (increasing or decreasing it depending on reached temperature and residence time, especially affecting sandy soils, Fig. 5), higher susceptibility to raindrop impact and loss of infiltration capacity when soil crusting or high water repellence are developed. When a temporal modification of the hydraulic soil properties is produced, runoff and soil erosion are enhanced and this produce on-site impoverishment of soil and off-site damages. Soil microflora and fauna may be directly affected by fires due to heat impact and the modification of the microhabitat, e.g. changes in soil physical and chemical properties such as pH or water content. As a consequence, the composition (more autotrophic and less symbiotic microbes), structure and functioning of the soil component results unbalanced and far from the pre-fire conditions, which is especially relevant for plant-microbe symbiotic associations (i.e. lower mycorrhizal formation). The microbial communities of the uppermost soil horizons may be more affected by fire than those from deeper horizons as heating extinct rapidly with depth. Recurrent fires increase nutrient losses that may not recover during the inter-fires period. Net reduction of soil fertility is the result of this imbalance.

Figure 5. Water repellence caused by fire on organic topsoil.

2.3 Effects on vegetation and fauna

Plant species of dry regions, including Mediterranean regions of the world, have developed many adaptations to fire along evolution. Mediterranean ecosystems usually regenerate efficiently after fire. Plant species recover either by resprouting (resprouters) (Fig. 6) or germination (seeders) (Fig. 7), or both (facultative seeders). Vegetation dominated by resprouters usually regenerate plant cover faster than vegetation dominated by seeders, therefore soil erosion and degradation hazard is lower in the former case. Seeder species germinate from the seed bank of the soil or the canopy (e.g. pines). Frequent fires in short intervals may prevent the recovery of the seed bank, and this is especially critical for pines. Mediterranean pines usually require 15-20 years to recover the seed bank of the canopy and they do not produce permanent seed bank in the soil. As a consequence, repeated fires in shorter intervals would make pines disappear on-site, and the recovery should come from nearby unburned stands. In case of large fires this would require long periods of time as pine colonisation rate is around 25 meters in 20 years. In general, endangered and rare Mediterranean plant species are not negatively affected by forest fires, even in some cases the contrary, unless fire is combined with other disturbances. In the same way, alien species do not generally proliferate in burned sites.

Fugure 6. Post-fire resprouting of Mastic tree (*Pistacia lentiscus*).

Figure 7. Mediterranean gorse is an obligate seeder that accumulates a high fine and dead fuel at maturity (12-18 years), produces high intensity fires and slowly regenerates from the seeds after fire.

The impact of fires on animals is very variable depending on their size and mobility, although generally faunal activity is drastically reduced after fire. The most affected groups are probably reptiles. Nesting birds are often selected as sensitive indicators of fire impacts in the fauna. Data from southern France show that full recovery of bird communities in forests may require 25-30 years. However, recently burned forests and shrublands usually develop good pasture, which improves herbivore's habitat and hunting potential.

2.4 Fire vulnerability.

Post-fire management requires the prediction of fire impacts on the ecosystems and landscapes, and their social and economic consequences.
Degraded vegetation because of long-term over-exploitation may have lost regeneration capacity, especially under frequent wildfires. Soil types show variable fragility in front of forest fires according to critical soil properties (erodibility, infiltration capacity), and climate and topography conditions.
Ecosystem vulnerability to wildfires can be assessed combining information on the recovery capacity of vegetation, on soil erodiblity, and on climate and topography.

3. The special case of wildland-urban interface

Land abandonment and the current spreading of settlements into the wildlands are dramatically increasing the wildland-urban intermix perimeter. This generates high ignition hazard from human activities to forest fuels, and high risk of forest fires to affect housings and other urban structures. Therefore, the growing wildland-urban interface presents a sharp increase of the risk of casualties. Some countries have developed regulations to reduce fuels in the perimeter of houses and to reduce flammability of building materials in houses.

4. Strategies to reduce fire hazard, mitigate fire impacts, and restore burned wildlands

4.1 Setting priorities

Management objectives for the wildlands could be manifold. In relation to the impact of forest fires, a minimum set of prioritised objectives should apply in most of the cases:
1) Soil protection and water regulation.

2) Reduce fire hazard and increase ecosystem and landscape resistance and resilience to forest fires.
3) Promote mature, diverse, and productive forests.

Depending on degradation risk and management objectives, several mitigation and restoration strategies might be applied (Fig. 8, Vallejo, 1996).

Figure 8. Strategies for post-fire restoration in the region of Valencia, Eastern Spain (Vallejo, 1996).

Development of criteria to identify vulnerable ecosystems Case study: Valencia region (Spain)

The vulnerability of an ecosystem is its degradation susceptibility in front of disturbances. It can also be defined as the inverse of the buffer capacity for disturbances without losing quality. This quality or value of the ecosystem may consider both the damages caused, and the devaluation of the resource, including the socio-economic (recreational, wood productivity) and ecological value (erosion, successional dynamics, landscape). Erosion, vegetation dynamics and landscape structure are the three components considered for the assessment of ecological vulnerability to fire, and the proposed methods intend to assess mainly the extent of ecosystem damage and the response capacity after a wildfire. The temporal scale runs from the very short term (< 1 yr, for the identification of erosion sensitive areas) to the medium term (25 yrs, which will reveal changes in vegetation composition and structure). Nevertheless, there are some relevant aspects that are little predictable such as fire intensity and post-fire meteorological conditions. Therefore the assessment should be completed by field or remote analysis of both parameters to orientate specific restoration actions.

Short-term assessment (< 1 year).

Bedrock and soil type, rainfall intensity, slope gradient and length, and erodibility of the mineral fraction of the soil are physical features that modulate the soil erosion risk, jointly with vegetation structure (Fig. 9). In the first year after the fire, the ecosystem capacity response will depend on the properties of the affected vegetation that determine the recovery rate at the short-term.

In general terms, the shorter the time needed by vegetation to cover a threshold percentage of the soil (30-40%) the lower the erosion susceptibility. Species vital attributes (resprouting ability, persistent seed bank, growth and dispersal capacity) may be used as a predictor of post-fire vegetation response. The rate of vegetation recovery will be the result of combining the reproductive strategy of the burned vegetation (seeder vs. resprouter species) with other physical factors such as climate and aspect. We assume that resprouting species recover after the fire faster than obligate seeders (Pausas & Vallejo, 1999) and the resprouting capacity does not depend on precipitation. On the contrary, the emergence of seedlings of obligate seeder species is highly dependent on rainfall in the next spring or autumn after fire.

Figure 9. Methodological approach for the assessment of short-term ecosystem vulnerability.

Medium-term assessment (≈ 25 years)

At the medium term, ecosystem vulnerability will depend on its ability to persist without major changes attending to composition, structure and relative species cover/biomass. In general, mature forests dominated by obligate seeder species, both in the over- and understory, show medium to high vulnerability depending on their germination capacity after fire. For instance, Aleppo and maritime pines (*Pinus halepensis* and *P. pinaster*) show high seed survival and germination rate after fire due to serotinity of their cones (Fig. 10). On the other hand, seeds of stone pine (*P. pinea*) and sub-Mediterranean or temperate pines (*P. nigra*, *P. sylvestris*) and junipers (*Juniperus phoenicea*) hardly germinate after fire, hampering vegetation recovery as it was before the fire occurred. Mature forests dominated by resprouting broadleaved species (i.e. hol oak *Quercus ilex*) are highly resilient (little vulnerable). Immature forests dominated by seeders are more vulnerable than mature ones, as pines do not massively produce viable seeds and hence recruitment is rather low. Finally, shrublands dominated by obligate seeders (*Ulex parviflorus*, *Cistus* spp., *Rosmarinus officinalis*) show medium vulnerability as they are, in general, well adapted to fire with great and highly dynamic soil seed banks. Moreover, their germination is stimulated by the fire and/or the new conditions generated by fire (e.g. more radiation reaching soil surface that produces huge daily temperature oscillation).

Figure 10. Aleppo pine (*Pinus halepensis*) regeneration: the passage of fire opens serotinous cones releasing shortly afterwards the seeds on the ashbed. With the first relevant rainfall in autumn or spring, a high germination is often produced taking advantage of the high nutrient availability and little plant competition.

Strategies and actions for post-fire ecosystem restoration and fire prevention

Case study: Ayora site (Valencia, Spain)

Soil type and land use history (often closely related), disturbance regime and topography features determine the type and characteristics of the regenerating community after fire. Fig. 11

reflects the observed and projected dynamics of vegetation following a wildfire in a mature pine forest, taking into account bedrock and the occurrence of further fires.

Figure 11. Conceptual model of the dynamics of Mediterranean vegetation after fire from a common pine forest starts. Bedrock (soil type) and fire recurrence are included in the model. Arrows are transitions from one vegetation type to another. Question marks indicate unknown transitions. Ule par = *Ulex parviflorus*; Ros off = *Rosmarinus officinalis*; Que coc = *Quercus coccifera*. From Baeza et al. (2007).

Forest ecosystem recovery can be very slow under Mediterranean conditions and intermediate stages of successional vegetation before arriving to a forest structure use to be very combustible. Most of these transition types are dense shrublands with a strong fine and dead fraction component. These characteristics confer a very high fire hazard to these formations, which facilitates the system to enter into fire degradation loops. Actions planned for these situations have to break the degradation cycle and promote new trajectories to get closer and faster to the target forest ecosystem. Actions may aim at the same time to reduce fire hazard. Within the EU-funded SPREAD project (*Forest Fire Spread Prevention and Mitigation*), the present experiment aimed to improve vegetation quality by reducing fire hazard and increasing vegetation resilience, and to break the positive feedbacks between fire and landscape homogenisation. Model ecosystem selected for this research was fire-prone senescent shrublands dominated by an obligate seeder, gorse *Ulex parviflorus* (Fig. 7), where woody resprouters (both shrubs and trees) were absent or scarce. Restoration techniques consisted in planting resprouter species and selective clearing of vegetation leaving the few standing individuals of resprouters and pines. Four experimental treatments were considered: control (shrubland), cleared, planted shrubland, and planted cleared plots. The clearing residues were chipped and applied as mulch.

Three years after the application of clearing, we observed a deep change in vegetation structure and fuel model (Fig. 12 & 13). Selective clearing shifted a highly flammable dense and continuous shrubland, with a great amount of dead biomass, to a grassland with sparse resprouting shrubs where fuel load is discontinuous. Obviously, biomass accumulation was significantly reduced from ca. 3,000 to 500 g m^{-2} but total plant cover only decreased from 85 to 56% and bare soil in the cleared plot was lower than 5% due to the slash layer protecting soil surface. The proportion of plant cover occupied by resprouter species in relation to that occupied by seeders greatly increased three years after the application of the clearing treatment (Fig. 14). The soil surface mulched with brush-chipping greatly reduced the germination rates of obligate seeders. Total number of seedlings m^{-2} in control plots was 2-fold higher than under the mulch layer. As a result, the resprouters vs. seeders rate in the cleared plots is 10-fold that of control ones, conferring a higher resistance and resilience capacity to the new ecosystem.

Figure 12. Physiognomy of a mature to senescent Mediterranean gorse shrubland.

Figure 13. Aspect of a mature to senescent Mediterranean gorse shrubland one year after clearing and planting with seedlings of resprouter species.

Figure 14. Effects of vegetation clearing on the plant cover percentage ratio of resprouter and seeder species three years after the application of clearing.

The survival and growth of the introduced seedlings was also sensitive to experimental treatments, especially to vegetation clearing. The plantation of late-successional broadleaved tree species in Mediterranean degraded areas has traditionally produced low survival and growth rates. It is then noteworthy the high survival rates of holm oak (*Quercus ilex*) and *Rhamnus alaternus* seedlings (around 90%) which are a success by itself. Introduced species improved seedling growth when vegetation was cleared.

In conclusion, the combination of clearing and planting resprouting species is a suitable option for managing fire-prone shrublands because fuel load and fuel build-up rate are drastically reduced and the introduced resprouting species confer high resilience to the ecosystem, decreasing ecosystem vulnerability to fire and, hence, to desertification.

5. Selected references.

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