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Long-term Restoration Strategies and Techniques

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Abstract

Long-term post-fire restoration not only aims to restore the ecosystem structure and function, but also endeavors to recover ecosystem fire resilience and reduce future fire propogation potential. This generally requires restoration strategies that promote secondary succession towards more mature, more resilient plant communities at a landscape scale. Pre-fire planning is essential to prioritize vulnerable sites and develop plans for these areas. Fire behavior models are often used for this process.

Current restoration techniques (plant species selection, seeding of woody plants, development of quality nursery stock, site preparation, soil amendment and fertilization, etc.) typical of the semi-arid Mediterranean environment are described with recent study results providing examples. However, the usefulness of these techniques is proven in the field where the complex interaction of long-term climate, short-term weather events, introduced plants, soil properties, extant organisms, etc. make each restoration project unique. Given the uncertainties of environmental conditions and the myriad of interactions, adaptive management principles should be applied to long-term post-fire restoration.

INTRODUCTION

In general, long-term forest fire impacts requiring restoration actions are caused by: a) wildfires affecting fire-sensitive ecosystems in regions where natural fires are uncommon; b) unprecedented fire frequency or severity (i.e., altered fire regime) over fire-dependent ecosystems; c) unprecedented combination of fire regime and other disturbances over fire-dependent ecosystems. For example, Mediterranean ecosystems can be considered fire-

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dependent/influenced as they have evolved under fire influence (Pausas and Verdú 2005, Pausas et al. 2006), and Mediterranean plants show a large array of adaptations to cope with fire impact (Pausas et al. 2004a). However, during the last decades, fire regimes have been deeply altered (Pausas and Vallejo 1999). This fact, in combination with other long-term anthropic disturbances, may cause further fire-induced degradation beyond the resilience domain of Mediterranean ecosystems.

The Mediterranean basin has been subjected to extensive and intensive exploitation for millennia (Vallejo et al. 2006). In many instances this exploitation has been excessive and resulted in land degradation. As far back as 2500 years BP, Plato complained about the degradation of upland forests and dramatic soil losses (Yassoglou 2000). As a consequence of this long-term human impact, most of the Mediterranean basin is now regarded as 'degraded' (TNC 2007), whereas most of the other Mediterranean-climate regions of the world have suffered less degradation. Therefore, fire impacts on ecosystems should be analyzed in terms of the interactions between direct fire-induced processes and previous human-induced degradation processes. And post-fire rehabilitation should include a long-term perspective on recuperating ecosystem integrity according to ecological restoration concepts (van Andel and Grootjans 2006). In addition, as fire hazards are inherent in the Mediterranean and other world ecosystems, fire prevention principles should be incorporated into post-fire rehabilitation strategies to reduce the number of future fire events.

Post-fire regeneration in fire-dependent ecosystems usually follows the autosuccession process, in which the same plant species composition and relative abundance regenerates after a fire (Trabaud 1994). However, this model does not always occur. There are several woody species that do not regenerate either after a single fire (Riera and Castell 1997, Retana et al. 2002) or after short fire intervals (e.g., *Pinus halepensis* and *P. pinaster*; Vallejo and Alloza 1998). In addition, post-fire weather conditions and/or seed bank exhaustion can drastically affect obligate seeder species regeneration (Faraco 1998, Baeza 2004).

This chapter will present the rationale for long-term post-fire restoration strategies and describe the techniques used. As most of the research in this area has been conducted in Mediterranean-climate and other dry regions of the world, and the Mediterranean basin in particular, these regions are the focus of the chapter.

POST-FIRE ECOSYSTEM RESILIENCE

Mature Mediterranean ecosystems are often dominated by shrub and tree species that have the ability to resprout after fire (and also after cutting and animal browsing). Resprouting plants quickly regenerate plant cover from below ground reserves, even in summer, quite independently of rainfall events (Vallejo and Alloza 1998). This trait of an ecosystem shows high resilience to wildfires (Ferran et al. 1992) as most of the pre-fire species reappear in similar density soon after a fire and soil protection is achieved rapidly – reducing the risk of increased runoff, soil erosion, and degradation (Abad et al. 1996).

Historically, Mediterranean ecosystems have been degraded by burning, crop abandonment, overgrazing, wood gathering, and charcoal production (which often involved uprooting the largest shrubs and trees), and these disturbances have been combined in multiple space and time sequences (Vallejo et al. 2006). However, in European Mediterranean countries, these practices have been strongly reduced in the last three to four decades through a generalized process of tertiarisation of rural economies. This is likely to occur in the near future in the Mediterranean countries of Africa. Thus, a generalized process of land abandonment has taken place since the 1960s in Europe, and is continuing under the Common Agricultural Policy.

Abandoned lands are colonized by opportunistic species, which in the early stages are mostly obligate seeders (Gallego et al. 2004). These species have short life cycles and many of them generate an abundant and persistent seed bank. Woody seeders are often strong fuel accumulators; they lead to high fuel load accumulation and thus to fire-prone shrublands (Baeza et al. 2006). In fact, the dramatic expansion of large wildfires in European Mediterranean countries has been partly attributed to the extensive land abandonment occurring in the region (Vallejo and Alloza 1998). Wildfires affecting fire-prone shrublands that have colonized old fields often enter into short-interval fire cycles that stop any further secondary succession towards more mature ecosystems. Even without fire, some early stages of secondary succession are stable and inhibit late-successional species colonization (Debussche et al. 1996). Many opportunistic shrubs have the ability to colonize both old fields and burned ecosystems (Baeza and Vallejo 2006). In the short-term, ecosystems dominated by obligate seeders regenerate slowly after fire, thus leaving bare soil exposed to wind and water erosion for relatively long periods of time (Vallejo 1999). This may result in irreversible soil degradation/ loss at the ecological scale and enhance the long-term, whole-ecosystem degradation that had started prior to the fire. Therefore, land abandonment promotes short-term fire cycles that result in ecosystem degradation loops. Recovering ecosystem resilience would thus require breaking these loops and promoting secondary succession towards more mature, more resilient plant communities (Vallejo and Alloza 1998).

Woody resprouters often produce big seeds, with dispersion mediated by animals (Pausas et al. 2004a). Some of these seeds show high water requirements for germination (Montoya 1993), and require the presence of dispersors (Alcantara et al. 1997). Their often fleshy and highly nutritional fruits are very attractive to predators (Waller 1993). All these circumstances place serious constraints on the ability of these species to colonize degraded sites (Laguna and Reyna 1990). Therefore, where natural colonization of late-successional woody resprouters is not sufficient, artificial introduction through seeding or plantation may be required to improve ecosystem resilience (Vallejo et al. 2006).

PLANNING POST-FIRE RESTORATION

Identification of Vulnerable Ecosystems

For fire-prone areas, preparedness must be incorporated into forest management planning. Post-fire rehabilitation and restoration measures require pre-fire planning to prioritize vulnerable sites and timely post-fire implementation of restoration actions. Assuming that the first objective in post-fire rehabilitation is the mitigation of runoff, flash floods, and soil erosion, early post-fire interventions have to be concentrated at the most vulnerable sites. These can be identified in a given area by using erosion models, basic cartography, and GIS (see Alloza and Vallejo 2006). When a fire occurs, emergency seeding and other techniques can be applied (see Chapters 10, 11, 12 and 13 in this book). Although early interventions may not be part of a long-term perspective on post-fire restoration, it is important to avoid early interventions that may work against the long-term plan. For example, if early interventions introduce alien herbaceous species, this might hinder the normal progression of recovery through secondary succession (see examples in Robichaud et al. 2000).

Longer-term restoration is appropriate when the existing vegetation shows low resilience to forest fires, loss of key forest species has occurred, and/or regeneration of fire-prone formations is likely to occur (Fig. 1). These vulnerable plant formations can be identified from vegetation cartography and/or forest inventories (Alloza and Vallejo 2006).

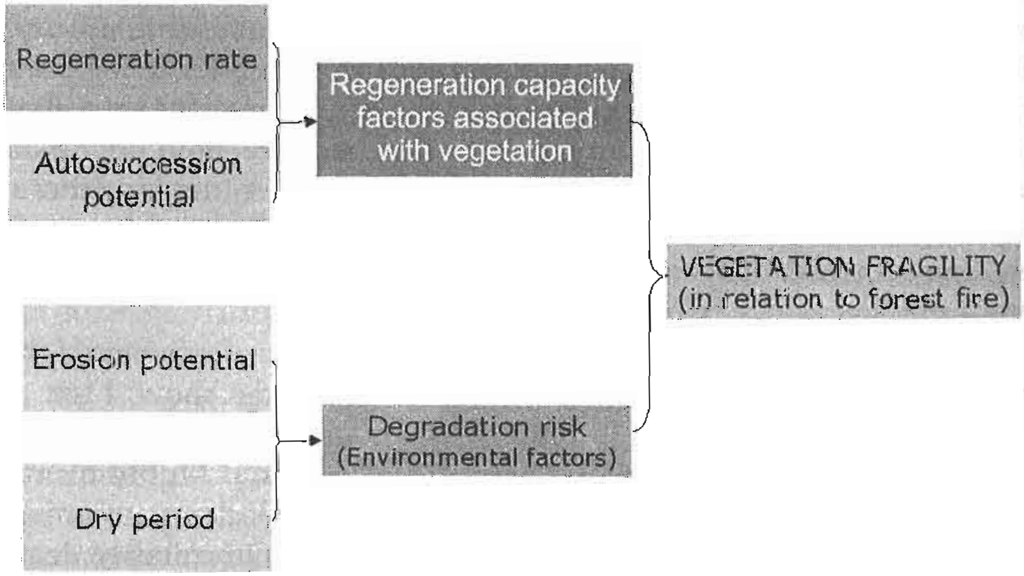

Fig. 1 Scheme for identifying ecosystems vulnerable to forest fires with the use of GIS.

Planning Restoration at the Landscape Scale

Planning restoration in fire-prone regions at the landscape scale should aim at reducing landscape combustibility. Traditionally, it is believed that disturbances are more likely to spread across a homogeneous area (Wiens et al. 1985), but the opposite also occurs (Turner 1987). It has been proposed that in highly fragmented landscapes disturbances require a higher boundary-crossing frequency and a more convoluted route and, therefore, spread less easily (Turner and Romme 1994, Forman 1995). In the case of fires, it is generally accepted that greater landscape heterogeneity retards fire propagation (Minnich 1983, Wiens et al. 1985, Knight 1987), although landscape pattern may have little influence on crown fire behavior when burning conditions are extreme (Turner et al. 1994, Keeley et al. 1999). No universal correlation has been found between fire propagation rate and landscape heterogeneity (Morvan et al. 1995). Landscape-scale fire patterns are the result of complex interactions among topography, weather and vegetation (fuel type, moisture, quantity, and spatial distribution) (Turner and Romme 1994, Hargrove et al. 2000). The topographic and physiographic features of the landscape influence the local probabilities of initial ignition and burning patterns, while the spatial arrangement of fuel categories also influences initial ignition as well as fire growth and behavior.

Large increases in fire occurrence were experienced in many of the Mediterranean areas in the 1970s due to land-use changes. Since the beginning of the 20th century, intensive land abandonment and decrease in grazing activities have generally resulted in increased fuel loads and expansion of large, interconnected non-wooded patches (Duguy 2003) throughout the ecosystem. The landscapes became highly fire-prone and the risk of large fires increased. The large number of fires that did occur generally caused further homogenization of these landscapes (Debussche et al. 1987, Vos 1993, Vázquez and Moreno 1998).

To reduce both fire occurrence and fire spread while promoting the expansion of a forest in the landscape, Forman and Collinge (1996) proposed three main approaches which focused on landscape pattern:

1) minimize the sites that are especially susceptible to fire ignition;
2) increase landscape spatial heterogeneity; and
3) increase barriers or filters that inhibit fire spread.

Using the FARSITE model (Finney 1998) for fire simulation, we determine fuel model distributions and fire-break networks that would reduce fire risk at the landscape level, and hence provided guidance for forest restoration aimed at fire prevention (Fig. 2; Duguy et al. 2005). FARSITE simulations showed that large interconnected patches of heavy surface fuels (mature dense shrublands) favored fast and intense fires. The fragmentation of this highly fire-prone matrix through the introduction of dense woodlands (i.e., the creation of a more fine-grained landscape *sensu*; Forman 1995) was very

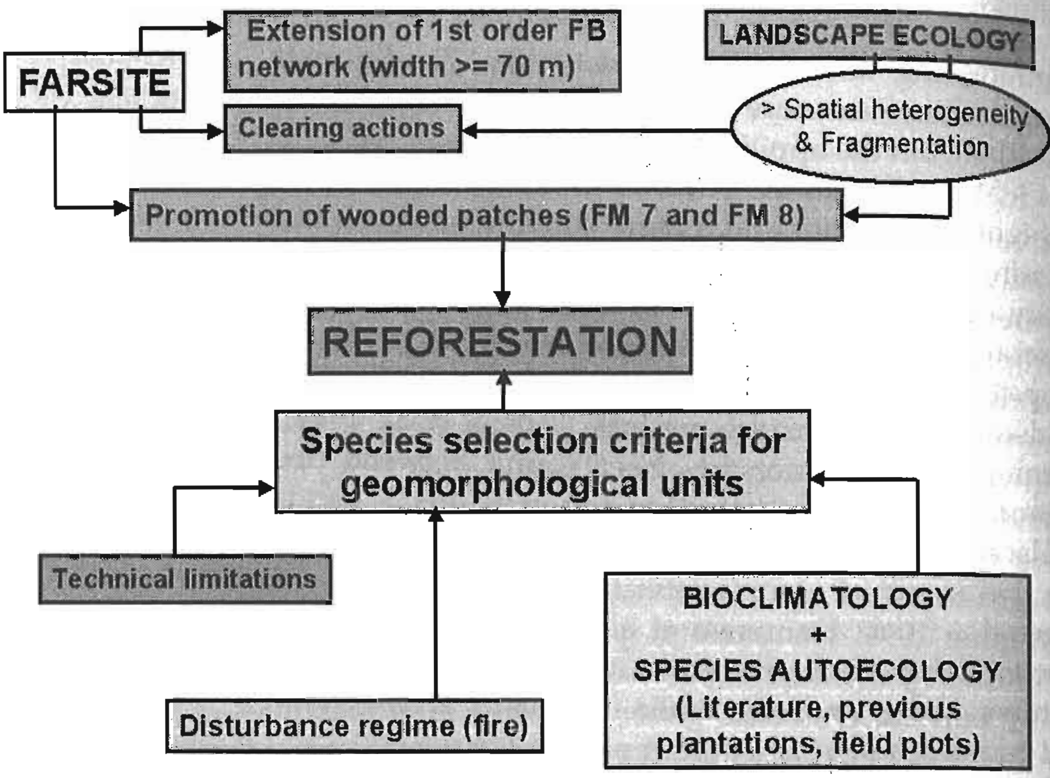

Fig. 2 Outline of the planning procedure for managing and restoring the landscape for fire prevention and landscape functional quality enhancement: fuel breaks (FB) design, fuel clearing treatments in forest to promote fuel models (FM) 8 and 9 (low combustibility) and reforestation.

effective in reducing the fire size and, in most cases, in reducing burning conditions (rate of spread, fireline intensity). Other effective landscape-level fuel alterations were the introduction of forest corridors between woodlands and the promotion of complex patches (high perimeter/area ratios or high fractal dimensions) among wooded patches. As these latter patches are potential sources for colonization processes, all actions that increase the edge length between wooded patches and non-wooded patches favor forest expansion. Surface fuel reduction actions applied over large areas (e.g., extensive clearing actions) were also an effective way of controlling fire spread, limiting fireline intensity, and lowering potential fire-caused damages (Byram 1959, Ryan and Noste 1985). Fuel reduction on fire-prone shrublands dominated by seeder species can be conducted in conjunction with plantations of woody resprouters (Baeza et al. 2005) to achieve the double benefit of reducing fire hazard and improving ecosystem resilience and diversity (see the section, *Plant Species Selection*, below). Our results also showed that similar degrees of fragmentation might lead to different fire sizes and fireline intensities, depending on the precise spatial arrangement of the various woodland successional stages.

It appears that a certain degree of heterogeneity and fragmentation of the vegetation structural diversity provides resistance to fire spread (Agee et al.

2000). It also provides a wider range of environmental resources and conditions, thereby promoting higher biodiversity in the landscape. Nevertheless, further research is still needed to identify the relationships between fuel models (used to determine fire growth and behavior and associated critical values of target landscape structures) and sustainable landscape management strategies. Coupling firebreak networks with appropriate landscape-level fuel treatments also seems to be a good strategy for limiting the occurrence of large, high-intensity fires, and thereby, reducing the associated negative effects on the ecosystems. However, fire management at the landscape scale may be very expensive, and a cost-benefit analysis would be needed, especially in areas where strong winds may reduce their effectiveness.

PLANT SPECIES SELECTION

In the last decades, species selection criteria in reforestation plans have been immersed in the native/exotic and conifer/hardwood discussion. The use of native flora is a priority in conservation-based reforestation (FAO 1989), but it has frequently been hampered by the limited success of seedling establishment. The success of exotic species may be related in many cases to the absence of specific pests, biogeographical isolation and, especially, to their early-successional features, although there is often a risk of either lack of adaptation or its opposite: extreme aggressiveness. Some of the native trees proposed for reforestation are late-successional, which makes them more sensitive to biotic and abiotic conditions (Hughes and Styles 1987, Zobel et al. 1987). This is the case of native tall shrubs and broad leaved resprouters. It is generally assumed that late successional species have low survival possibilities when introduced in open or degraded lands, although this is not always so (Ashby 1987). In the case of Mediterranean forests, attempts to introduce *Quercus* spp. seedlings commonly faced high mortality rates, making this alternative very expensive (Mesón and Montoya 1993). In addition, for historical and biological reasons, the techniques for introducing late-successional native species are poorly developed (Zobel et al. 1987), and up to a few years ago the scarcity of native plant material available from nurseries was a barrier to diversifying restoration practices. The forestry tradition uses conifers as pioneers to restore degraded lands, and after some years of silvicultural treatments, hardwoods are introduced under the pine canopy in an improved soil and microclimate conditions (see, for example, Montero and Alcanda 1993). Nevertheless, young pine plantations are very vulnerable to fire and the use of pines alone in reforestations is especially risky in wildfire 'hot spots'. Recent advances in the ecophysiology of hardwoods offer much-improved seedling plantation results on open degraded lands in the Mediterranean (Baeza et al. 2005).

In the context of ecological restoration, plant species selection for post-fire recovery of the ecosystem (i.e., structure and function), reduction of future fire risk, and improving fire resilience involves the following considerations (Fig. 3; Vallejo and Alloza 2004):

- The first step is to determine what native species are suitable for restoring the habitat. This is not straightforward in extremely degraded ecosystems. Often, remnants of the (supposedly) original vegetation are used as a reference after phytosociology investigations. This should be regarded, however, as a broad indication only. The presence of a species close to the site to be restored, under similar physiographic conditions, is a more reasonable indicator of species compatibility with the habitat since soil degradation may have made the habitat unsuitable for the reference species. More direct information is provided by auto-ecological studies, but these are still scarce for many native species of potential interest. This

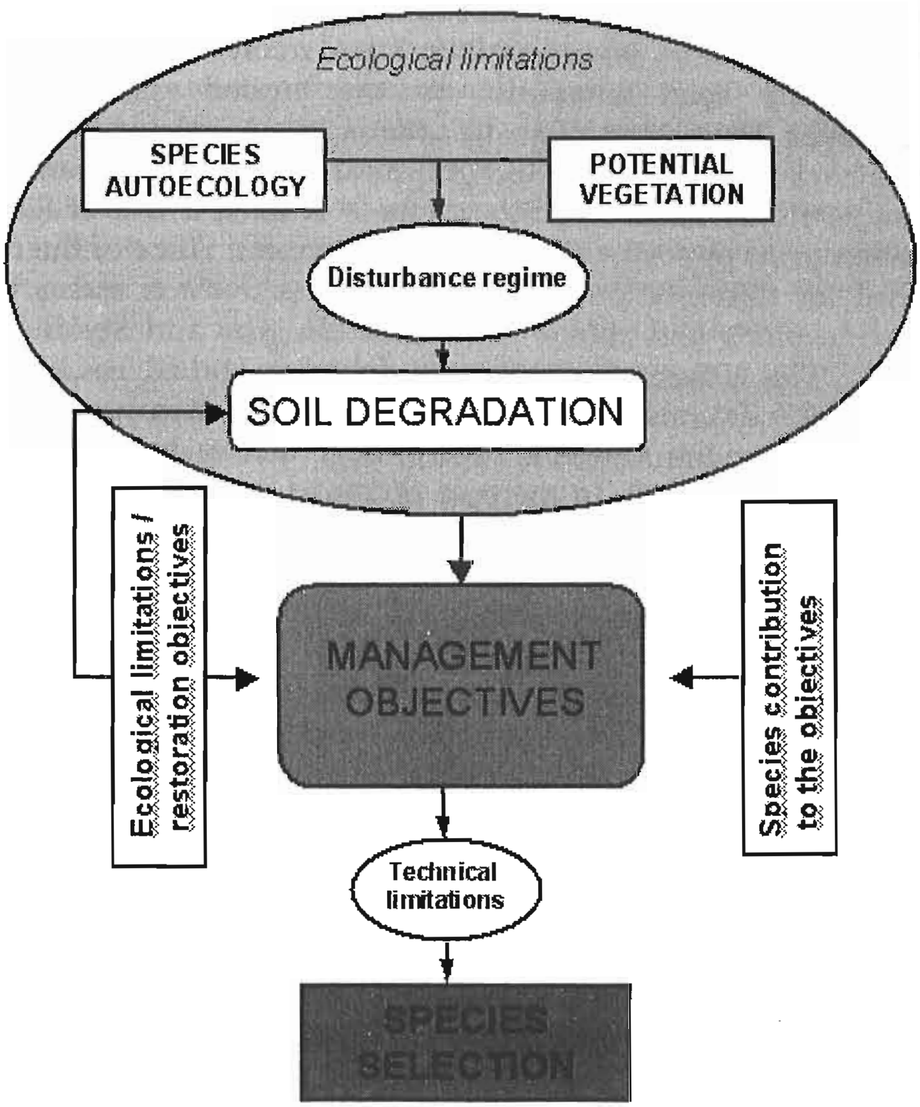

Fig. 3 Species selection constraints and criteria for afforestation.

area deserves further research. The plant species selected have to be adapted not only to the climate (including extreme events) and soil conditions, but also to the prevailing perturbation regime (e.g., wildfires and pests). For example, pine species were routinely used in afforestation actions in the past, both in the Mediterranean and in many other regions of the world. Most pines do not survive fire intervals that are shorter than the time period required to produce enough seeds, which is around 15 to 20 years in the Mediterranean (Pausas et al. 2004b).

- From the set of species found to be suitable for restoring a given habitat, the species that best fit the management objectives should be selected. In the case of post-fire restoration we would select woody resprouters according to the above-stated objectives of increasing fire ecosystem resilience and reducing fire risk. Resistance to fire, defined at the individual species level, should be related to species flammability, which is determined by plant structure (fuel density and size), necromass proportion, moisture content, and the presence of components that enhance or diminish flammability (volatile organics, resins). At the community level, resistance should be related to the combustibility of the ecosystem, including species composition, structure of the stand and characteristics of the litter bed. For example, in the Mediterranean, *Ulex parviflorus* is considered highly flammable, especially in mature stands that accumulate a lot of standing dead fuel; *Quercus coccifera, Erica multiflora, Rhamnus lycioides* and *Juniperus oxycedrus* are considered to show medium flammability; and *Pistacia lentiscus* and *Rhamnus alaternus* show low flammability (Elvira and Hernando 1989). Considering fuel loading, especially fine and dead fuel, and surface/volume ratio, Papió and Trabaud (1991) found that *Pistacia lentiscus* presented low fire hazard, whereas *Genista scorpius* (with a similar structure as *Ulex parviflorus*) presented high fire hazard. Trabaud (1976) emphasized the role of the litter layer in the combustibility of Aleppo pine (*Pinus halepensis*) forests, more than the flammability of the pine species itself. Other objectives might be considered, such as improving soil fertility through introducing N-fixers (Binkley and Giardina 1998) or enhancing carbon sequestration (Lal 1999).
- Restoration usually consists of introducing one or several keystone species. These species, typically trees or tall shrubs, are supposed to play a critical role in determining ecosystem structure and functioning, acting as 'ecosystem engineers' (Jones et al. 1994) that modify the habitat. It is assumed that these species will improve soil properties, create a forest floor habitat, improve the microclimate, and indirectly facilitate the importation of seeds by birds. Finally, the introduction of a woody species could not be enough for its complete establishment if symbionts, pollinators, or dispersers are lacking (Hobbs and Norton 1996). Mycorrhiza and/or rhizobateria inoculation in the nursery is a way to

ensure efficient symbiosis for seedlings to be introduced (Barea and Honrubia 2004).

- Technical constraints may impede the introduction of a specific species in a restoration project. Adequate technical knowledge of species cultivation requirements and plantation techniques are essential for successful introduction. Species growing in the same habitat may show contrasting growth and physiological strategies (Vilagrosa et al. 2003a, Vilagrosa et al. 2005), and hence may require different cultivation techniques in the nursery. In addition, this basic ecophysiological knowledge is very limited for many of the most promising species for restoration worldwide and especially in tropical regions. The main environmental limitation for a successful introduction of plants on degraded Mediterranean sites is water stress, and this is, of course, also applicable to other arid regions of the world. In Mediterranean regions, the most critical situations are located in the transition between semi-arid and dry sub humid climates, where high water stress is combined with high disturbances, especially fire.
- Finally, cost constraints always limit the practice of restoration and its innovation.

SEEDING WOODY SPECIES

For afforestation, seeding may offer many advantages over planting, especially in time and cost savings. It has also been suggested that seeding is easier to mechanize and reduces the risk of root deformation. Seedlings developed directly on site are expected to acclimatize better to the site conditions from the early plant development phases. However, the unreliability of this direct seeding method, which yields inconsistent seedling emergence and survival and growth rates is the main reason for its limited use (Winsa and Bergstern 1994).

In general, seeding techniques include: a) row seeding (sowing seeds in strips across an area); b) spot seeding (dropping a number of seeds on a small spot to ensure the emergence of at least one seedling in each spot); and c) broadcast seeding (scattering seeds over the entire reforestation area; Barnett and Baker 1991). The first two techniques include soil preparation and seed covering with a thin soil layer that facilitates seed germination by keeping soil moisture around the seed. In addition, covering the seeds with soil may reduce seed predation because it makes them undetectable for visually-searching seed-predators and may also limit detectability for predators that use olfaction to find seeds (Nilson and Kjältén 2003). Broadcast seeding offers fewer guarantees for germination, unless a mulch layer is applied after seeding; it does represent, however, the most rapid and inexpensive seeding technique. Other specific techniques have been developed including micro-site preparation (Bergstern 1988, Wennström et al. 1999), which consists of sowing seeds in small inverted pyramidal indentations that increase seed moisture by improving soil-seed contact and reducing evaporation.

Aerial seeding, a type of broadcast seeding, has been used since the 1950s and involves the dropping of seeds from helicopters or fixed wing aircraft. The most important advantage of aerial seeding is its potential to seed remote areas with limited access and to treat large areas in a short time and at a low cost. On a calm day under optimum conditions, a helicopter can cover up to 1200 ha; however, the usual daily average is about 600 to 800 ha (Barnett and Baker 1991). Aerial seeding may be appropriate in circumstances where the previous vegetation has been removed, such as after a fire, extensive logging, or in reclamation of a mine site. It has been widely used for emergency seeding after fire (see Chapter 11 on non-native and native seeding in this book) in western USA, and as a supplement to natural regeneration after logging in northern Europe, USA, and Canada. Some attempts at pine forest restoration after fire using aerial pine seeding have been made in eastern Spain (Peman and Navarro 1998); however, only the results of a single experiment conducted in the Sierra del Garraf (Barcelona) have been published to date (Castell and Castelló 1996). In this study, 2 kg ha^{-1} of *Pinus halepensis* seeds mixed with 18 kg ha^{-1} of inert wheat (added to ensure better pine seed distribution and to provide predator satiation) were sowed. A relatively successful average pine germination of 5 percent, representing an overall density of 6000 seedlings per hectare, was reported. Nevertheless, the results were highly variable depending on the terrain (from 12,000 seedlings per hectare in old fields to seven-times-lower densities on hillslopes and at the bottom of valleys). These reasonably good results were probably related to mild temperatures and with abundant rainfall that occurred just after seeding.

Seed predation is one of the major causes of direct seeding failure. After a recent fire in the Valencia region, seed predation was assessed in a *Pinus halepensis* aerial seeding project completed in late November. In four of the six monitored plots, more than 80 percent of the pine seeds were predated in two months, and at the end of spring no germinated pine was found (Pausas et al. 2004c). Therefore seeding success may depend, at least in part, on reducing the seed predation.

Correct timing of seeding and appropriate use of seed pre-treatments to overcome seed dormancy may be critical for taking advantage of the most favorable conditions for germination and thus ensuring rapid germination. In a study to determine the effects of seed priming on subsequent *Pinus halepensis* seed germination in the field, seeds were primed for 6 days at 20°C in sand moistened with a 10^{-3} M gibberelline solution. The primed seeds germinated earlier than the control seeds, and in some cases, the prime seeds had higher germination rates (Fig. 4). The effectiveness of seed priming may vary with climatic conditions. In this experiment, seed priming was most effective in suboptimal temperature conditions (i.e., late autumn). Seed pre-treatment may be useful to ensure rapid germination when seeding is conducted in autumn.

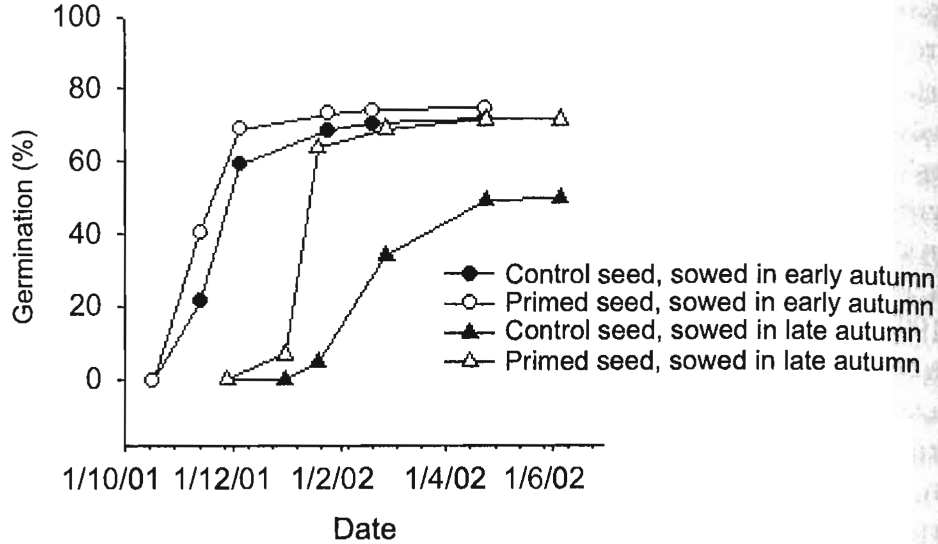

Fig. 4 Cumulative percent germination of primed and control seeds of *Pinus halepensis* sowed in early or late autumn. Experimental seeding was conducted in Alicante (eastern Spain), using pots placed outdoors with moderate watering.

PLANTATIONS

Plant Quality: Nursery Cultivation

Seedling plantations on drylands and degraded soils are often discouraging because of high mortality rates and poor growth. In general, climatic conditions after planting are one of the major limiting factors for seedling establishment. Suitable restoration techniques may help the seedlings to get through the transplant shock and first summer drought, and establish successfully. These include several nursery techniques that take into account the morpho-functional characteristics of seedlings to promote their resistance to drought conditions and increase their acclimation to the reforestation site. The main technical elements in the nursery culture are:

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

Substrates or growing media

The characteristics of the growing media are important for good root development—a key step in the success of a plantation (Peñuelas and Ocaña 1996). The recommended growing media includes standard components, such as peat moss or other organic materials (coconut fiber, composted sawdust, bark, or composted sewage sludge) in combination with aeration materials (e.g., perlite, sand, vermiculite, tuff, or polystyrene; Landis et al. 1990). Several decades ago, foresters thought that the use of raw substrates based on topsoil produced better rustic plants that were well adapted to harsh field conditions. Natural topsoil is difficult to standardize; not only is it very heavy, which hinders planting operations, but also it often comes from excavations for constructions and has poor fertility. Our experiments in eastern Spain using different types of growing media showed that those based on topsoil produced poor results in terms of survival and growth (Fig. 5). A mixture with small amounts of hydrogels or some clays (sepiolite) can increase the water holding capacity of the plug, thus providing the seedlings with high water availability for a longer period of time (Fig. 6). This can be especially important in semi-arid climates with high rainfall variability.

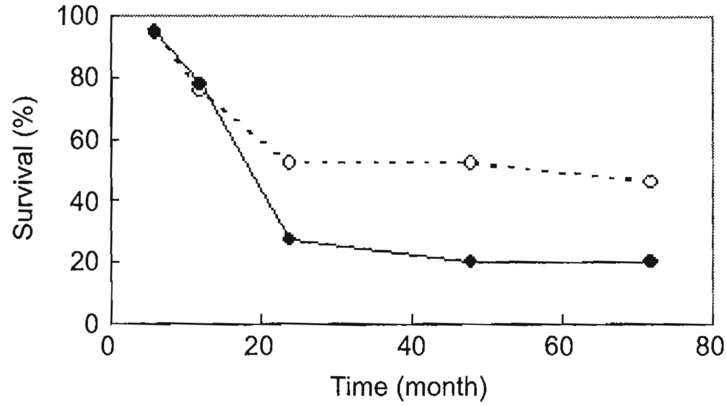

Fig. 5 Field survival dynamics of *Pinus halepensis* seedlings cultivated in two different growing media: compacted low quality topsoil with sand (black circles), and high quality topsoil with peat (white circles).

Containers and root systems

Several studies have related the planting stock quality of the seedlings to the type of container used (Landis et al. 1990, Peñuelas 1995, Vilagrosa et al. 1997, Dominguez et al. 1999). An appropriate container should have a shape and dimension that allow the seedling to develop correctly, especially its root system. In past decades, seedlings were grown in pots and polyethylene bags that often produced deformations in the root system, like taproot spiralling and/or reduced lateral root growth (Peñuelas and Ocaña 1996, Vilagrosa et al. 1997). Recently, producers tend to use pipe-shaped containers suspended in the air with channels or ribs inside them. This type of container prevents taproot spiralling by facilitating aerial root pruning, which in turn favors the development of secondary roots. Moreover, the interior channels or ribs promote the downward growth of roots and avoid spiralling. Nevertheless, it is difficult to find a universally acceptable type of container because the container must be adapted to many factors, such as species, nursery management, and planting needs. In general, high-volume containers (300 cm^3 or more) are recommended for reforestations in dry and semi-arid climates and for species with high root-to-shoot ratio, because they allow for the critical root system development during the first stages after planting. In our experience, long containers are preferred for species that develop a tap root,

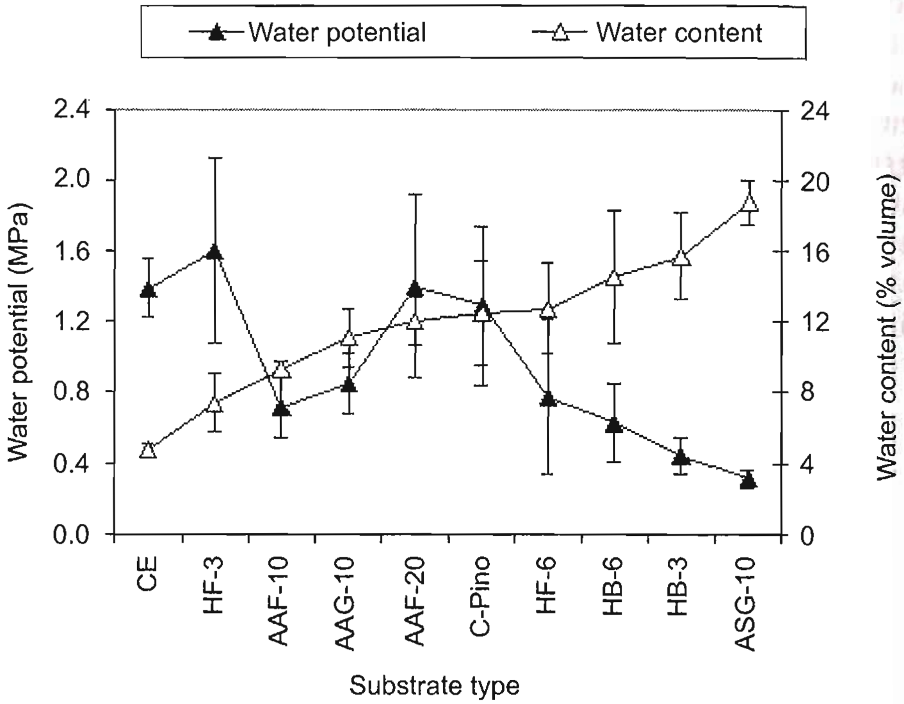

Fig. 6 Comparison of different substrate type during a drought period (Chirino and Vilagrosa, unpublished data). After 7 days of drought, substrate mixed with (ASG-10) maintained high water content and high predawn water potential compared to the control substrate (CE). The mixture with hydrogels (HB-3, HB-6, and HF-6) improved the water-holding capacity of the substrate when compared with the control substrate. Substrate type: CE substrate mixed with composted pine bark 25% (C-Pino); hydrogel Bures at 3 and 6% (HB-3 and HB-6); hydrogel Stockosorb at 3 and 6% (HF-3 and HF-6); coarse clay Sepiolite at 10% (ASG-10); fine clay Attapulgite 20/70 at 10 and 20% (AAF-10 and AAF-20); and coarse Attapulgite 4/20 at 10% (AAG-10).

like *Quercus* sp., while wider containers are recommended for species that show important secondary root development.

Drought preconditioning

There is a great deal of evidence indicating that a major obstacle to plantation success is transplant shock, that is, the intense short-term stress experienced by seedlings when they are transferred from favorable nursery conditions to the more adverse field environment. Drought preconditioning, the induction of drought resistance mechanisms, is one of the main techniques used to prepare seedlings for drought stress (Landis et al. 1998). However, a characteristic of arid region plant species is their ontogenetically high resistance to stress conditions. The most commonly used drought-preconditioning techniques (i.e., short-term preconditioning), designed for plant species characteristics in humid or subhumid climates, are of little benefit when applied to dry-climate species (Fonseca 1999, Vilagrosa et al. 2003b). For dryland species, long-term drought preconditioning in the nursery promotes greater benefits to plant morpho-functional characteristics than short-term preconditioning (Rubio et al. 2001, Chirino et al. 2003). On the other hand, the response to drought preconditioning seems to depend on the plant species. For example, species like *Pistacia lentiscus* are very responsive to preconditioning while species like *Quercus coccifera* are not. The kind of response is probably related to the drought strategy developed by each species (Vilagrosa et al. 2003b).

The main responses obtained in drought preconditioning experiments are: higher root-shoot ratio in the nursery (Chirino et al. 2003), changes in allocation patterns (i.e., higher fine root colonization in the plantation hole and lower above-ground development; Fonseca 1999, Rubio et al. 2001, Chirino et al. 2003), higher tolerance to drought conditions by means of higher elasticity of the cell membrane (Rubio et al. 2001) or better photochemical efficiency (Vilagrosa et al. 2003b), and drought-avoidance mechanisms such as higher root hydraulic conductivity for supplying water to leaves, higher leaf capacitance to water, and lower transpiration rates (Villar-Salvador et al. 1999, Vilagrosa et al. 2003b). In general, drought preconditioning does not improve survival, but it produces healthier seedling in field conditions (Rubio et al. 2001).

Fertilization

Given that nutritional status affects basic morphological and physiological plant processes, fertilization influences seedling growth and development. In the last decades, forest seedling fertilization practices have moved from using the lowest fertilization rates possible to maximize hardening of the seedling, to the current strategy of increasing fertilization to produce a seedling that can resist stress with sufficient photosynthetic capacity and carbohydrate reserves to initiate vigorous growth in the field. Recent studies indicate that larger, well-fertilized seedlings respond to field conditions better than smaller, less-fertilized seedlings (Villar-Salvador et al. 2000, Puértolas et al. 2003). Similarly, a positive relationship has been observed between survival or growth and nitrogen content in leaves (Oliet et al. 1997, Puértolas et al. 2003). However, under more limiting environments (semi-arid climate, irregular rainfalls) these results might not apply. Trubat et al. (2004), analyzing a wide range of species in semi-arid climates, observed that in years with a scarcity of rainfall, the bigger and better fertilized seedlings showed higher mortality rates than the smaller, less fertilized seedlings. Root growth potential was not promoted by higher fertilization, and seedlings were observed to develop root biomass according to their initial size (Trubat et al. 2004).

Field Testing

Recent findings on seedling quality have stressed the importance of promoting morpho-functional characteristics acclimated to a target ecosystem.

Doing so will reduce variability in seedling success. However, acclimatizing to a target ecosystem means that seedling quality cannot be determined at the nursery alone; it must to be tested in the field.

Site Preparation

Site preparation for reforestation generates a certain degree of soil disturbance, which may temporarily increase the risk of soil erosion (Shakesby et al. 1994). Thus, it is recommended that soil preparation work for plantations be applied at least two years after a fire, when the soil is less vulnerable and the regenerated plant cover provides a minimum protective threshold. The objective of site preparation is to increase the effective soil volume for root growth, to improve the capture of runoff, and to increase the soil water-holding capacity, in order to enhance seedling survival in the short-term. Due to its suitability for steep slopes, pit planting is a commonly used spot-treatment in soils with abundant rock outcrops, or in degraded areas where the existing vegetation can play an important role both in the recovery process and in soil conservation. When no machinery is employed, its effectiveness in increasing the soil water-holding capacity is low due to the small volume of soil affected by this technique. However, the small disturbance is a positive feature in terms of reducing the risk of soil erosion (Alloza 2003), preserving the specific richness and woody seedlings density, and reducing possible damage to the natural standing vegetation. Linear subsoiling is one of the most widely used soil preparation techniques, and it generally yields higher seedling growth and survival than spot treatments (Espelta et al. 2003, Bocio et al. 2004). This method provides a higher volume of effective soil for root growth, and a higher water-holding capacity. On the other hand, it may increase soil erosion and negatively affect the visual impact on the landscape, especially in rocky soils.

Water availability is the main factor hampering ecosystem restoration in dry or semi-arid areas (Vallejo et al. 2000). Current techniques that increase the amount of water available in the planting hole are: the application of different inorganic (hydrogels; Hüttermann et al. 1999) or organic amendments (composted or uncomposted refuses; Querejeta et al. 2000) or the construction of small water-harvesting structures associated with the planting holes (micro-catchments; De Simón 1990, Fuentes et al. 2004). The micro-cachment technique involves dividing the slope into several units that reduce its length and, as a consequence, the erosive strength of the runoff water. This soil preparation includes the excavation of shallow furrows to collect the runoff water in the plantation hole, and the excavation of a bench with a ridge to retain water. An inaccurate procedure or the occurrence of extreme rainfall events may generate the breakdown of the structure, leading to concentrated runoff and rill erosion.

Soil Amendments

Shallow soils or soils with poor structure may need high nutrient pools to maintain an acceptable seedling performance, and fertilization may compensate for these physical drawbacks. Planting holes may benefit from the application of biosolids, which act as a slow-release fertilizer and can provide longer-lasting effects than inorganic fertilizers. Additionally, biosolids promote microbial activity and increase the soil water-holding capacity and infiltration rates, resulting in higher water availability for the target seedlings. The negative effects of biosolid application are related to increased salinity and, if using semi-liquid sludges (slurry), changes in soil physical properties that occur as the sludge dries. Determining the optimum application rate is the key to this technique. Some studies suggest that doses of 15 to 30 Mg (dry weight) ha^{-1} are best for a *Pinus halepensis* plantation under dry-sub-humid Mediterranean conditions (Valdecantos et al. 2004). Using composted biosolids as mulch in the restoration of semi-arid open shrublands has proven to be effective in increasing the soil microbial functional diversity based on the local microflora.

Hydrophilic gels are synthetic products with the ability to absorb and retain high amounts of water in relation to the volume they occupy, thereby increasing the soil water-holding capacity (Hüttermann et al. 1999). The hydrogels retain water at a high matric potential, so it is easily available to the root. Once applied to the soil, the moisture in the rhizosphere lasts longer. In addition, hydrogels may incorporate some fertilizer properties or even fungal inoculum (Mikkelsen 1994). Applications of these products change the soil structure by modifying the size of soil aggregates and porosity, which implies an improvement in water storage capacity, soil aeration, and drainage. Hence, hydrogels can reduce transplant shock during the short time when the seedling roots are within the zone of the hydrogel influence. Nevertheless, in areas where the water deficit is extremely high, applying these polymers to the soil may result in negative consequences for the target seedling due to the high affinity of the hydrogel for the small amount of available water. In loamy or finer textured soils, the hydrogel moisture is subjected to suction by the clays under drying processes, thus reducing its chances of being used by the roots. Therefore, the positive effects of hydrogels are likely to be more relevant in sandy soils than in finer textured soils.

Tree Shelters

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

Most tested species, including some growing under Mediterranean-humid conditions, showed a positive response to tree shelters (Costello et al. 1996). Nevertheless, several experiments in dry Mediterranean conditions showed that tree shelters did not improve the overall survival, even though positive interactions were found among tree shelter treatments, site conditions, and species (Vilagrosa 2001). A detailed analysis revealed that, in terms of survival, the effect of tree shelters was more important in the driest regions. The implementation of treeshelters is especially recommended in restorations that involve introduction of pre-germinated acorns that may be predated in high percentage by small rodents (Seva et al. 2004).

In relation to growth, tree shelters mainly improve stem elongation (Rey Benayas 1998, Domínguez et al. 1999, Cortina et al. 2004, Seva et al. 2004). The main effects reported from the use of tree shelters involve reductions in both the water deficit and the incoming radiation (Kjelgren and Rupp 1997, Kjelgren et al. 1997). These conditions would favor the development of morpho-functional traits of shade-tolerant plants: stem elongation, larger leaves with lower specific leaf weight, higher chlorophyll content, higher shoot weight ratio, etc. (Kozlowski et al. 1991). Despite the fact that these traits may seem negative for the survival of introduced seedlings, the lower radiation and higher relative humidity inside the tree shelter may favor more efficient photosynthetic machinery and lower transpiration rates, thus increasing water-use efficiency.

One of the main problems described for unventilated tree shelters (i.e., those with no lateral holes) was the increased temperature inside, which may be deleterious for seedling growth and survival (Burger et al. 1992, Bergez and Dupraz 1997). However, in ventilated tree shelters, temperature changes were minimal (Seva and Cortina 1999).

CONCLUSION

At present, strategies and techniques are available to address the long-term ecological restoration of degraded ecosystems/landscapes after wildfires. However, the restoration process is subject to many uncertainties that cannot be foreseen and will undoubtedly affect the success of restoration. Of course, most restoration efforts involve vegetation enhancements that are very dependent on both long term climate (wet-dry cycles) as well as short term weather events (adequate rainfall for germination), both of which are uncertain. In addition, detailed knowledge of the ecosystem to be restored and the potential interactions between introduced plants, soil properties, extant organisms, etc. may also be incomplete and add to the uncertainties. Therefore, restoration projects should follow adaptive management principles

(Whisenant 1999), including monitoring and project modification as circumstances change over time. Although adaptive management will lead to greater restoration success, this dynamic approach requires more time and longer-term funding than usual projects that do not include monitoring or adaptation actions.

Long-term post-fire restoration is an expensive process that should be clearly justified in terms of improving landscape and ecosystems quality (biodiversity, resilience, structure, function, etc.) and reducing wildfire propagation. Therefore, quality control and evaluation should be incorporated in the design and budget of a restoration project (Vallauri et al. 2005, www.ceam.es/reaction). Unfortunately, from a public perception point of view, the results of restoration actions are not immediately apparent making it difficult to justify the expense to the public. Long-term demonstration projects may be used to showcase the value of long-term restoration activities.

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