RESTORATION OF MEDITERRANEAN-TYPE WOODLANDS AND SHRUBLANDS

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11.1   INTRODUCTION

There are only five Mediterranean-climate regions (MCRs) on Earth – the Mediterranean Basin, most of California, central Chile, southern South Africa and south-central and south-western Australia. They all have cool, or cold and relatively wet winters alternating with long, hot and dry summers. Spring and autumn seasons are ephemeral in comparison, and highly variable. During much of the year, droughts lasting weeks or even months are frequent, with grave consequences on all biota, since water is the key limiting factor for growth and well-being of all organisms, including humans. Yet, biodiversity is unusually high in all five MCRs. Well-adapted woodlands and shrublands, which are the subject of this chapter, are numerous and noteworthy, in ecological, cultural and socio-economic terms.

Climate, geology and evolutionary biogeography of MCRs have all contributed to alpha, beta and gamma biodiversity of these regions. In the Mediterranean Basin, California and Chile, the predominantly young, orogenic systems produce intense geomorphologic dynamics with poorly developed, shallow soils (Bradbury 1981). The uneven relief with steep slopes in large parts of the territory and spreading of unconsolidated and soft substrata increase the risk of soil degradation. In sharp contrast, MCR landscapes in South Africa and Australia are ancient and stable; as a result, they have highly weathered and leached soils that are very poor in nutrients (Rundel 1998). Human land use histories have also contributed to ecological diversity in all five regions, and must be taken into account when contemplating or undertaking ecological restoration.

Natural *wildfires* are common in most of the five MCRs, owing to the high accumulation of fuels leading to enhanced flammability in summer, the frequency of lightning storms and, in some areas, periodic but intense, hot and dry winds such as the Santa Ana in California and the Mistral in southern France. An exception is central Chile where the Andean Cordillera protects the MCR area from summer storms and lightning (Rundel 1998). As a result, central Chilean sclerophyllous vegetation is ill adapted to frequent fires, never having had selective pressure from this form of disturbance over evolutionary time. Nonetheless, in recent decades anthropogenic fires have become much more common, and this is causing profound changes in the characteristic features of central Chilean landscapes (Armesto *et al.* 2009).

In spite of the similarities in climate, many striking differences can be found among MCRs, in addition to fire regime. As mentioned, historical differences in *land use* practices, including the time span of degradation processes, socio-economic dependence on local resources and cultural perceptions of the relationship between humans and nature, all have great impact on biota and ecosystem dynamics. The Mediterranean Basin is the only MCR in the so-called Old World, where humans have practised agriculture for as long as 13 millennia in some areas (Purugganan & Fuller 2009). The process, initiated in the Near East, is thought to have spread to the entire Mediterranean region over the course of about three millennia (Zeder 2008). In the process, people consumed resources and transformed, or 'resculpted' natural **landscapes** and **ecosystems** to their own ends (Blondel *et al.* 2010). Therefore, environmental **degradation** in the region is ancient (Thirgood 1981). However, the intentional fires, herbivory by domestic livestock, agroforestry practices and constant trade of wild and domestic species of plants and animals first developed in the Mediterranean Basin itself, are now common practices there, and indeed in most of the MCRs (Armesto *et al.* 2007). They cannot be described simply as environmental destruction. Instead, in more neutral terms, we may refer to the profound transformation of the 'original' or pre-Holocene landscapes (Blondel *et al.* 2010). Certainly, the combination of the great diversity of physical conditions (geology, topography and soils) and of land use histories, and finally the large number of possible pathways and stages of succession occurring after various disturbances all give to Mediterranean landscapes a particular patchwork pattern and kaleidoscopic quality that Blondel and Aronson (1999) called a 'moving mosaic'. Both the interpretation of these landscapes, and the procedures for carrying out ecological restoration there, are highly complex.

In the four MCRs other than the Mediterranean Basin, intentional anthropogenic transformation of landscapes was less intense and extensive until very recently. However, the indigenous cultures of each four of those regions also have a long history of landscape modification, mostly through the intentional use of fire. Natural fire regimes were often altered, affecting ecosystems dynamics, landscape structure and biodiversity in the process (Pausas & Keeley 2009).

Of course, urbanization, industrialization and 'metropolization', and the globalization of commodity and service markets, are driving current land uses and

cultures to higher convergence in all five MCRs, just as elsewhere, but still the 'fingerprint' of land use history is strong enough to influence current restoration approaches, of which there are a large number indeed in the various MCRs. As elsewhere, the first step towards restoration is to identify and halt degradation processes. Such processes affecting MCRs today are land degradation produced by long-term overuse of natural resources, and further degradation produced in case of land abandonment in semi-arid areas, anthropogenic forest fires and the spread of invasive species. Once one or more ecological **thresholds** have been crossed, however, certain degradation processes may not be reversed spontaneously, but only through human **intervention** in the form of restoration actions and manipulations. In particular, in recent decades, anthropogenic fires have increased in frequency and intensity, not only in Chile (see above) but also in the other MCRs. This represents a serious threat to ecosystems, wild biota and, of course, the built environment. To cope with fire-induced damages to ecosystems and to humans, *post-fire rehabilitation* started in California in the 1930s and is now being applied in the other MCRs as well. Later in this chapter (section 11.2.5), we will dwell on the topic of fire regime intervention and restoration in some detail.

Since the mid-nineteenth century, several countries in the Mediterranean Basin have carried out **afforestation** programmes to protect watershed headwaters areas, regulate streamflow, reduce flash floods, control soil erosion and provide forest products. Programmes in Spain (Martínez García *et al.* 1996) and France (Vallauri *et al.* 2002) were particularly extensive. In contrast, afforestation in central Chile only began in the mid-twentieth century, and there top priority was given to timber production, and most often the use of exotic trees was favoured over native forest restoration (Becerra & Bustamante 2009). In south-western Australia and South Africa, afforestation and restoration practices have also begun in the recent past, but generally speaking the strongest emphasis of restorationists in those two MCRs has been put on the control of invasive alien species (e.g. Richardson & van Wilgen 2004), as is now the case in California as well (E.B. Allen *et al.* 2005; D'Antonio & Chambers 2006).

The rapid increase of global trade and travel in recent decades has facilitated the spread of invasive organisms of all kinds. In the case of the MCRs, the predominant direction of invasion has been from the Mediterranean Basin to the other four regions, and there have also been major invasions of alien plants from Australia into South Africa (see also Chapter 20). Negative effects of **invasive species** on Mediterranean ecosystems include reduced biodiversity changes in soil properties that make the site unsuitable for native species (Yelenik *et al.* 2004), and more frequent fire following annual grass invasions (Minnich 2008). However, biological invasions of both plants and animals are now clearly on the rise in the Mediterranean region as well, and much more attention is being accorded with this problem, by ecologists, horticulturists and conservation authorities (Quézel *et al.* 1990; Filippi & Aronson 2011a,b).

11.2 RESTORATION OF MEDITERRANEAN-TYPE WOODLANDS AND SHRUBLANDS

In this section, after discussing restoration priorities (section 11.2.1), we review some of the most prominent tools used in ecological restoration programmes in the range of ecosystems covered in this chapter. These include (1) post-fire emergency rehabilitation techniques such as sowing of herbaceous cover crops and mulching (section 11.2.2), and techniques to ameliorate reintroduction of key native tree species such as microsite preparation, tree shelters and bird perches (section 11.2.3), (2) invasive plant species control (section 11.2.4) and (3) the restoration of natural or moderate, cultural fire regimes (section 11.2.5). Approaches and techniques vary from region to region. We shall only explicitly consider **rehabilitation** activities as part of early post-fire measures, although they have short-term objectives not always related to the *recovery* of reference or pre-disturbance ecosystems. Furthermore, we will concentrate on the Mediterranean Basin, Chile and California. For more information on current restoration activities in Australian and South African MCR woodlands and shrublands, see Chapters 4 and 20, respectively.

11.2.1 Restoration priorities

The specific objectives of restoration differ widely among the different Mediterranean-type **woodland** and **shrubland** ecosystems, and are determined by the degree of degradation, and by climatic, biotic and socio-economic constraints (Table 11.1). As elsewhere

Table 11.1 Framework for the restoration of Mediterranean ecosystems. Drivers for restoration are identified, as well as actions that can be undertaken to attenuate them, and available techniques to implement these actions. Each driver must be offset to ensure successful restoration.

Driver	Action	Technique
Persistent anthropogenic disturbances, unwanted species	Release disturbance	Limited access to people, herbivores, etc.
		Fire prevention, windbreaks
		Species control (fire, herbicides, clearing)
Low propagule availability	Artificial introduction	Seeding, planting
	Promote dispersion	Bird-mediated restoration, frugivory-mediated restoration (artificial perches, catches and habitat amelioration)
Adverse environmental conditions	Reduce soil losses	Emergency seeding, mulching, sediment traps
	Ameliorate soil properties	Amendments, nutrient immobilization, mulching, drainage, soil preparation
	Improve microclimate	Shelters, mulching, microsite selection

in the Old World, the great and continuous density of historical layers in the Mediterranean Basin renders difficult the selection and use of a precise historical **reference system**, such as is frequently sought in New World settings (Egan & Howell 2001). References for ecological restoration are usually taken from pre-disturbance ecosystem information, in the case of recently disturbed ecosystems (e.g. due to fires), and from historical information or from reference ecosystems supposed to be natural, remaining in well-preserved sites. Sustainable seminatural ecosystems of socio-economic and cultural interest may well be taken as references for restoration projects, especially in areas like the Mediterranean region and highly transformed parts of the other four MCRs. In this case, as always, goals must be clearly defined in relation to site-specific constraints and opportunities. A good example is the *dehesa* or *montado* woodlands in Spain, Portugal and the rest of western Mediterranean Basin, which are artificially opened and managed savanna-like landscapes designed and maintained for silvopastoral or agrosilvopastoral uses (Aronson *et al.* 2009). In other MCRs, landscapes and disturbance regimes existing before the European settlements in the sixteenth–nineteenth centuries very often guide restoration actions.

Despite the wide range of MCR restoration objectives, some features are common to all MCRs:

1. Soil and water conservation is generally the main priority, for reducing and preventing soil losses and for regulating water and nutrient fluxes (Cortina & Vallejo 1999).

2. Increasing ecosystem resilience to disturbances, and ensuring the sustainability of restored lands by promoting the re-assemblage of plant, animal and microbial communities resilient to current and future disturbance regimes.

3. Improving landscape quality, from a local, cultural perspective, and the provision of ecosystem services (Figure 11.1).

4. Promoting biodiversity, fostering the reintroduction of key native species, while eradicating alien invasive species, and battling their re-establishment.

11.2.2 Early post-fire rehabilitation

In some cases, quick action is urgently needed before a degradation process reaches or exceeds a certain threshold beyond which restoration becomes prohibitively expensive or impossible. This may be the case in some fire-sensitive systems, where wildfire removes vegetation cover, leaving an unprotected soil. In eastern Spain, for example, many plant communities dominated by obligate seeders (species unable to resprout after fire), growing preferentially on soft bedrocks and, especially, on equator-facing slopes, commonly show slow post-fire *recovery* (Pausas *et al.* 1999) and thus high **erosion** and **runoff** risk. Early post-fire restoration can either be integrated in an ecological restoration programme, or just concentrate in short-term non-ecological objectives (e.g. plantations for timber production). Emergency action after wildfires has been practised in California for nearly a century, primarily to mitigate flooding and erosion. Federally mandated and financed post-fire activities in California, and throughout the United States, was formalized in 1974 through the Burned Area Emergency Rehabilitation
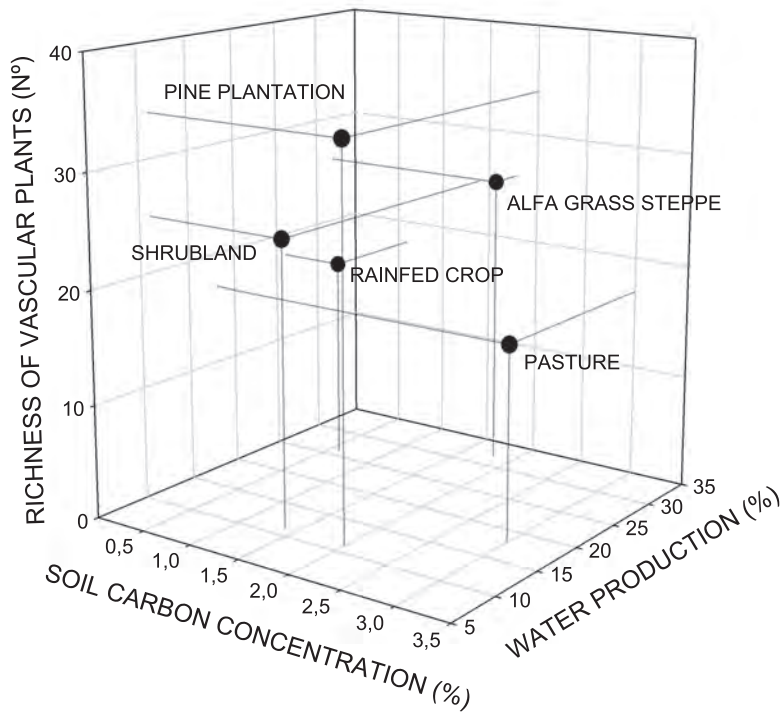
Figure 11.1 Indicators for ecosystem services provided by alternative land uses: soil carbon concentration for carbon sequestration, richness of vascular plants for biodiversity and water production for water regulation. This information may help evaluating restoration options. Average level of soil organic carbon (0–20 cm depth), richness of vascular plants (from floristic inventories) and water production (precipitation minus evapotranspiration and interception) in landscape units from a semi-arid area in south-eastern Spain (Derak 2010). Alfa grass: *Stipa tenacissima*.

programmes, now called Burn Area Emergency Response. In the Mediterranean Basin, there is no specific, nation-wide regulation for early post-fire rehabilitation; instead decisions are taken at the management project level.

To provide rapid soil protection, two main (non-exclusive) techniques are used: (1) *seeding* with herbaceous species, often including fast-growing, non-native species, and (2) *mulching*, that is, protection of soil surface with various kinds of organic materials. Seeding and mulching can be applied on large scales and in remote areas by using aerial means. Both seeding and mulching may reduce soil losses, surface crusting and water evaporation, and enhance water infiltration (Bautista *et al.* 2009).

There are also many examples recorded of the low efficiency of post-fire seeding on soil protection as a result of poor germination. Further, regrowth of native plants is often inhibited when introduced species have much higher germination rates (Beyers 2009). In the framework of ecological restoration, seeding would be only justified on degraded ecosystems with high post-fire degradation risk, showing low short-term regeneration capacity, and when using native species that combine both highly enough regeneration rate and low competition with late-successional species (Bautista *et al.* 2009).

11.2.3 Reintroduction of woody species

In many degraded lands affected by fires and/or other disturbances, it is necessary to **reintroduce** key **native woody species** to foster and accelerate succes-

sion (Vallejo *et al*. 2009). To ensure successful establishment, selection of woody species for restoration should be based, as much as possible, on the natural flora and vegetation of the area, and on the specific biophysical characteristics of the site. Traditionally, pine trees were planted in many areas in the Mediterranean Basin for catchment protection and sand dune stabilization. Pines commonly have high survival rates, allowing a relatively quick **revegetation** success. However, extensive pine plantations also provide an excellent fuel bed for large, devastating fires. Pine woodlands also have low **resistance** and **resilience** in the face of recurrent fires, and Mediterranean pines do not resprout after fire. On the contrary, most native hardwoods and, in general, woody sclerophyllous trees and shrubs resprout (Paula *et al*. 2009). Therefore, combining pines and hardwoods would take advantage of the complementary features of both groups of species, for example the high survival and fast growth of pines and the high fire resilience – that is, efficient resprouting capacity – of oaks (Pausas *et al*. 2004). Early attempts to introduce broad-leaved resprouting species in the Mediterranean Basin (e.g. *Quercus* spp.) failed because of high seedling mortality and, even until the late twentieth century, techniques for introducing these species in Mediterranean conditions were poorly developed. Since the last few decades, the introduction of hardwoods is becoming common in afforestation programmes, although the degree of success is still variable. Recently, **afforestation** objectives are widening their scope, so the diversity of introduced species is increasing in MCRs. New programmes even consider shrubland restoration, such as the restoration and rehabilitation of mixed *espinales* in a central Chile programme led by Carlos Ovalle of the Instituto Nacional de Investigaciones Agropecuarias. That long-standing programme is developing management and intervention techniques useful in restoring the structure and former levels of diversity and productivity of the mixed, anthropogenic *Acacia caven* formation (Ovalle *et al*. 1999). Therefore, new species, never tested before, including shrubs, are becoming of interest for restoration.

The reintroduction of hardwoods and other native woody species in degraded lands would require improved restoration techniques, in order to increase their degree of success. Drought is the critical factor hindering seedling survival under Mediterranean conditions. However, irrigation is seldom used in restoration projects of southern European countries or indeed

Table 11.2 Mediterranean restoration techniques concerned with water.

Objective	Technique
Increase water-use efficiency	Selection of drought-tolerant species and ecotypes
	Seedling preconditioning
	Improve below-ground performance
	Improve nutritional status
Increase water supply	Soil preparation and amendment
	Irrigation
	Microsite selection
Reduce water losses	Tree shelters
	Mulching
	Microsite selection
	Control of competing species

any of the MCRs. In central Chile, for example, where summer drought is particularly long and severe, all available water is reserved for other uses, mostly agriculture and tourism. In general, in the MCRs, as in all semi-arid and arid regions, the implementation costs of watering restoration sites in rugged and remote areas are prohibitively high. Therefore, as irrigation is not feasible, in most cases, improving rainfall capture and *water-use efficiency* become key factors for plant survival and growth. Two common practices in this regard are seedling manipulation and site preparation practices (Table 11.2). As elsewhere, species assemblages and nursery and field techniques must be tailored to different landscapes and landscape units, and restoration strategies should make use of available resources and functional processes remaining in the degraded site.

As most woody late-successional species do not form a permanent seed bank (Huston & Smith 1987), seeding woody species is an attractive technique to reintroduce target species owing to its low cost, the low impact of field operations and the improved possibilities for treating remote areas through aerial seeding. However, high predation risk and uncertain seed germination and seedling establishment under dry conditions are serious managerial constraints for the direct use of seeding (Vallejo *et al*. 2009). From several pine-seeding experiments carried out in Spain, right after major fires, only one produced acceptable seed germination rate and seedling survival, despite very favourable weather conditions (Pausas *et al*. 2004).

Seedling quality

How to characterize seedling quality is subject of much debate. Seedlings should be able to withstand unfavourable growing conditions (transplant shock, seasonal water stress and drought cycles), and take advantage of short favourable climatic periods to achieve sustained growth. Seedling quality has been substantially improved over the past decade (Cortina *et al.* 2006). Most approaches are based on nursery manipulations simultaneously affecting several plant morpho-functional traits. Seedling size together with other, mostly visually assessed, characteristics are used to define acceptable stocks. However, above-ground size may not be a good indicator of seedling quality under dryland conditions. Indeed, the relationship between seedling morpho-functional traits and establishment success is not clear, as seedling fate may be ultimately driven by the complex set of interactions allowing fast and deep rooting. The few long-term studies undertaken to date suggest that the positive effects of enhanced seedling quality may endure for a decade after planting (Cortina *et al.* 2006). Fortunately, many sophisticated *nursery* practices are now available to manipulate seedling traits, although there is still much scope for improvement in this area, particularly through the use of alternative, biodegradable and recycled materials as containers, as well as substitutes for standard soil mixes. Already, however, seedling quality is probably no longer a major limiting factor for the establishment of common species in the Mediterranean and in the other MCRs, provided that current knowledge is applied.

Microsite selection and nurse plants

The recognition of the importance of microhabitat heterogeneity – particularly the role of positive plant–plant interactions and the role of 'safe sites' – has contributed to the elaboration of a new paradigm for ecological restoration in MCRs. Facilitative interactions are especially important under semi-arid conditions, where isolated vegetation patches act as 'resource islands', mainly in terms of shade and soil fertility (Maestre & Cortina 2004). *Facilitation* may show a unimodal response to *stress*, with a maximum at intermediate levels and declining both at the lowest and highest levels of stress (Maestre & Cortina 2004), but the relationship between facilitation and stress is not simple. The patterned landscape of semi-arid

tussock steppes (*Stipa tenacissima*), common in southern Spain and throughout northern Africa, is a clear example of this, as tussock microsites have higher organic matter, higher water availability, lower temperatures and lower soil penetration resistance than intertussock patches. These environmental modifications facilitate the development of bryo-lichenic communities, and introduced woody plants (Maestre *et al.* 2003a). In other systems, the nurse plant can be a spiny shrub, as it protects planted seedlings from grazing (Gómez *et al.* 2001). In the Sierra Nevada, southern Spain, 4 years after planting, seedlings of *Quercus*, *Pinus* and *Acer* spp. planted under shrubs had, on average, rates of survival three times higher than those planted in open microsites (Castro *et al.* 2002). In contrast, under semi-arid conditions, *Pinus halepensis* may not be capable of facilitating the establishment of woody shrubs (Maestre *et al.* 2003b). The outcome of plant–plant interactions depends on the level of stress and on the morpho-functional attributes of benefactor and beneficiary species. Uncertainty in the results of plant–plant interactions represents a major challenge for the use of facilitative interactions in restoration programmes.

Heterogeneity in biotic and abiotic conditions in dry areas may be relevant to restoration success. Large-scale changes in slope aspect and bedrock can substantially affect the outcome of restoration. Further, subtle small-scale changes in soil properties and microtopography can also be important. Patchiness of plant population survival and persistence in apparently homogeneous areas are frequently associated with small differences in soil moisture, soil depth, stoniness, texture or nutrient availability (Figure 11.2). As plant responses to these factors tend to be nonlinear, it is very important to identify thresholds that may explain such differences in plant performance. A wide array of ecotechnological tools and techniques, which in many cases mimic abiotic conditions and biotic interactions, is currently available (see below). But it is important to note that the effect of local biotic and abiotic conditions on plant performance is commonly much greater than the effects of the various ecotechnological tools employed by restorationists (Cortina *et al.* 2011).

Soil preparation and amendment

Various soil preparation techniques have been developed to improve water supply to planted seedlings and ameliorate soil physico-chemical properties in degraded
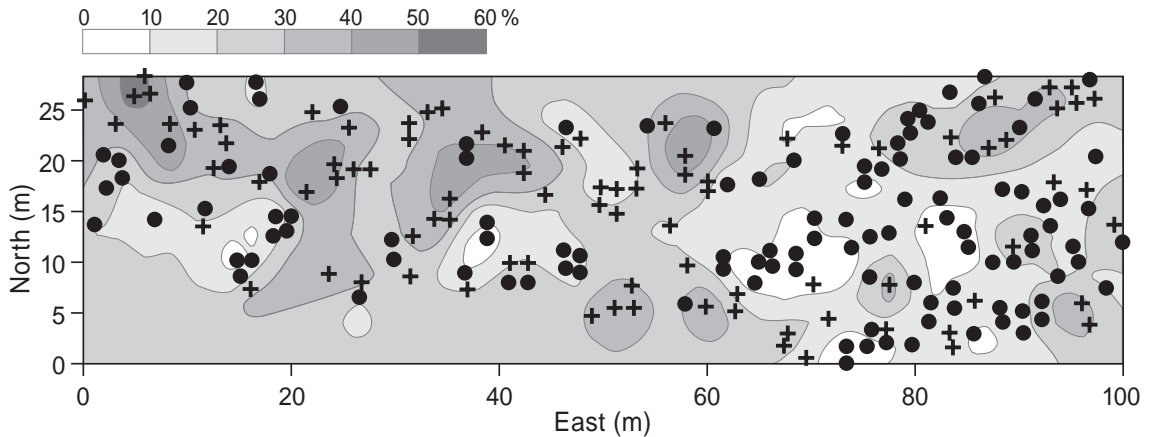
Figure 11.2 Small-scale spatial distribution of the amount of bare soil covering planting holes (as shown by the grey scale, in %; darker areas had more bare soil), and of seedling survival 1 year after planting. Crosses (+) and circles (•) are dead and live seedlings, respectively. There is a significant negative relationship between the amount of bare soil and survival (logistic regression; $P < 0.001$). (Elaborated from Maestre *et al.* 2003a.)

soils. The array of available techniques is very large. **Runoff** harvesting aims to intercept runoff and redirect water to the planted seedling. Successful results have been obtained in arid areas (Bainbridge 2007a; Figure 11.3), where low-infiltration surfaces allow runoff concentration in vegetated patches and increase the productivity of the whole system. In the example of Figure 11.3 (Fuentes *et al.* 2004), water harvesting increased seedling survival for the most drought-sensitive species assayed (i.e. *Quercus ilex*) and growth rate for the most tolerant (i.e. *Pinus halepensis*), although significant differences between species, in both variables, still remained. Therefore, the impact of additional water inputs provided by water harvesting would affect seedling survival right after outplanting under the most extreme dry conditions and for the most sensitive species. This is critical for the potential success of the plantation project, as seedling survival is not granted for many species and drought conditions (see Figure 11.5). Soil depth is very often a major limiting factor for seedling survival as shallow soils have low water holding capacity. Indeed, increasing planting hole depth from 40 to 60 cm increases seedling performance by 15% in eastern Spain, and the survival rate of planted woody species is very low for soils shallower than 40 cm in this region (Alloza 2003).

Mediterranean soils are frequently poor in soil organic matter and low in phosphorus availability (Vallejo *et al.* 1999). To what extent soil nutrient impoverishment is hampering restoration is a matter of discussion. Soil organic matter intervenes in many soil processes affecting plant growth, but especially in soil structure (i.e. stability versus soil crusting and erosion, and water-holding capacity) and soil fertility.

Positive response to the addition of resources (e.g. water, nutrients, or both) is considered an **indicator** that something was limiting. Planted seedlings commonly respond well to inorganic and organic fertilizers, and negative and null responses are associated with metal toxicity, salinity and increased above- and below-ground competition (Valdecantos *et al.* 2002). There is a wide range of organic residues available for improving soil fertility.

Artificial tree shelters

Artificial tree shelters are used to modify the physical environment of planted tree seedlings, acting effectively as mini-greenhouses. If properly designed, they can help reduce seedling transpiration and improve overall performance (Bellot *et al.* 2002), as well as provide protection against herbivory. The use of tree shelters has gradually increased, being readily adopted by land owners and other practitioners. Ventilated tree shelters help avoid excessive warming, while improving seedling survival and growth, and increase
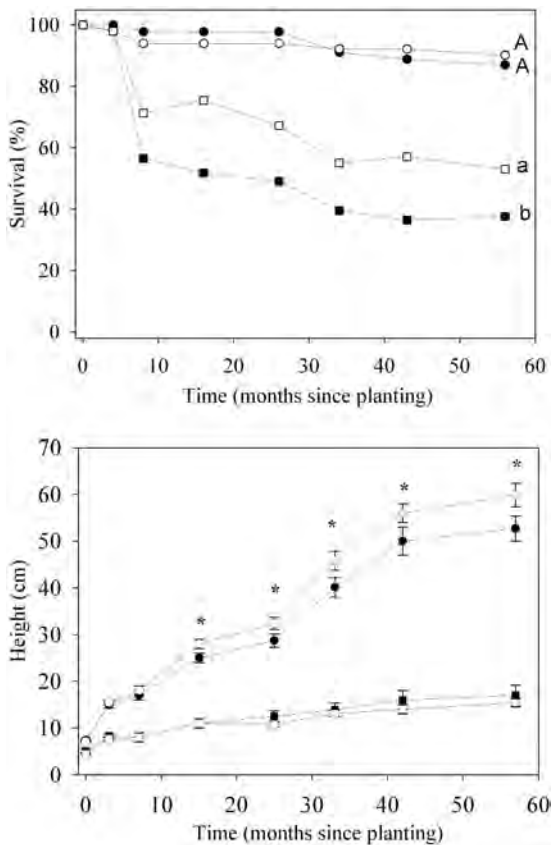
Figure 11.3 Survival and growth of Aleppo pine (*Pinus halepensis*, circle) and holm oak (*Quercus ilex*, square) seedlings in planting holes with (white) and without (black) water-harvesting microcatchments during the first year after outplanting. Field experimental data on degraded forests in the Region of Valencia (Spain). Different uppercase or lowercase letters within each species mean significant differences in survival between treatments (P < 0.05). For height growth, differences are indicated with *. (Modified from Fuentes *et al.* 2004.)

average stem size (Bellot *et al.* 2002; Figure 11.4). Nevertheless, the use of this technique alone does not guarantee plantation success. The root–shoot ratio of protected seedlings is often lower inside tree shelters than in unprotected seedlings, and this reduces the capacity of seedlings to withstand prolonged drought. Tree shelters significantly increase seedling growth in height with, however, the risk of producing elongated, 'leggy' stems. However, thanks to their slow growth

rates, most Mediterranean tree seedlings growing inside tree shelters gradually acclimatize to adverse climatic conditions outside the shelter and improve shoot growth. When this does not happen, unbalanced growth in stem height may occur, and the plant becomes highly susceptible to wind damage.

Costs and benefits of ecotechnological investments

Between 1992 and 2010, CEAM Foundation and University of Alicante (Spain) established 217 experimental plantations on degraded semi-arid and dry sub-humid areas. Initially, plantations used prevailing technology at that time, that is, using a few species (mostly pines and a small number of holm oaks), and simple nursery cultivation techniques and basic soil preparation. Over the past decade, plantations have incorporated innovations in species selection, nursery production, soil preparation and tending (e.g. mulches, tree shelters, organic amendments etc.), according to the discussion in previous sections. Seedling survival after the first summer in the field (a major indicator of plantation success) was strongly dependent on the number of consecutive days with no significant rainfall (precipitation events <5 mm) in both old and new plantations (Figure 11.5). But the slope of the relationship between survival and drought length decreased as advanced technologies were implemented. Under severe drought, survival rate would double, and the risk of total failure would decrease significantly by using advanced ecotechnology. One obvious interpretation of these results is that plantations carried out on harsh sites would require higher technological inputs and investments to achieve a given target than those on less harsh sites.

Bird-mediated restoration

The role of plant–animal mutualisms has been suggested for cost-efficient restoration plans (Handel 1997). Bird-mediated restoration is based on the observation of succession of abandoned woody crops, such as olive groves, carob groves, almond groves and vineyards. In these contexts, natural succession is significantly faster than in nonwoody crop fields thanks to the role of the trees as perch sites for frugivorous birds. These birds discharge seeds of late-successional species that germinate around the perching tree forming a nucleus of intermediate or advanced succession

Figure 11.4 Seedling height and diameter (mean and standard error) of holm oak (*Quercus ilex*) seedlings nine years after planting. CS: control seedling, S + TS: seedling with tree shelter, S + HG: seedling with hydrogel. The plantation was conducted in burned degraded forests in the Region of Valencia (Spain). Different letter means significant differences among treatments at the $P < 0.05$ level. Notice the low stem growth rate of holm oak in these degraded lands, on equator-facing slopes and under dry climate (average annual precipitation for the monitoring period: 360 mm). (Modified from Chirino *et al.* 2009.)
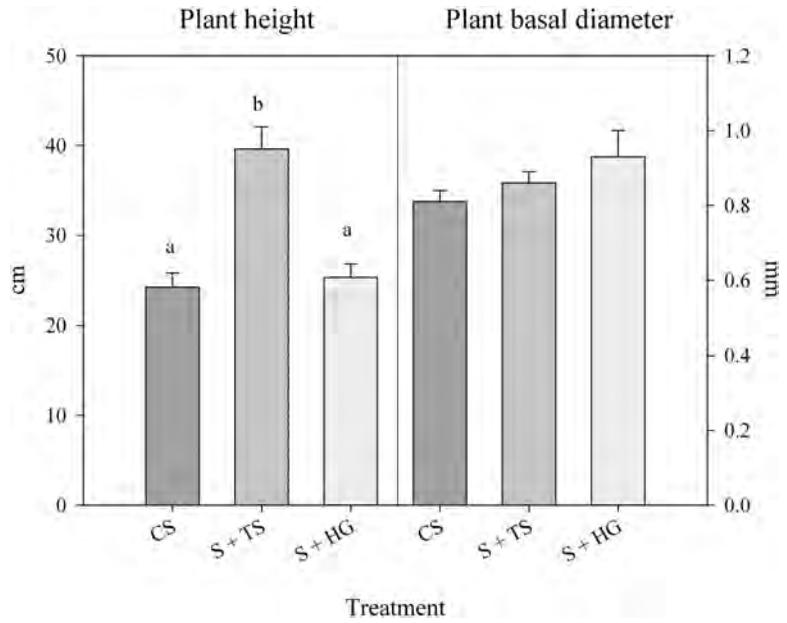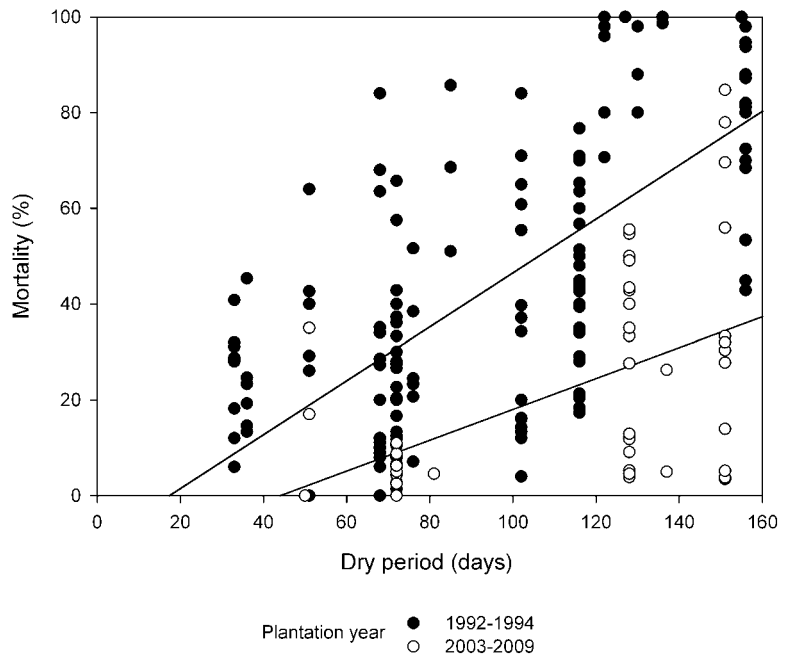
Figure 11.5 Percentage of plants surviving in relation to the length of the dry period during the first post-plantation year, for several native species planted in eastern Spain. The first set of plots (1992–1994) includes plantations using conventional techniques at those times. The second set (2003–2009) includes plantations using recent technical innovations. Regression lines for each plantation period are significantly different (ANCOVA significant interaction between plantation year and dry period, $P < 0.0001$). 'Dry period' refers to the maximum number of consecutive days with precipitation less than 5 mm. See the text for more details. (Elaborated from the CEAM-University of Alicante database. J.A. Alloza unpublished data.)

taxa (Pausas *et al.* 2006). In fact, these birds act as **ecosystem engineers**, *sensu* Jones *et al.* (1994). Furthermore, the perch tree may also create a favourable microsite for germination and survival. Dead trees may also be used by birds as caches (e.g. for acorns). These bird-mediated facilitation processes inspired a restoration technique based on providing bird perches (dead trees and artificial woody structures) in old-field sites to accelerate colonization rates and ecosystem restoration. Although being an attractive and inexpensive technique to help succession, it has seldom been applied to Mediterranean ecosystems and most examples come from tropical forest ecosystem restoration projects (see Chapter 9). In many areas of extensive old-fields, where there is generally a very low seed availability of late-successional species, this technique could be especially appropriate.

11.2.4 Principles of control for invasive alien plants in MCRs

The principal goal of restoration after invasive species control is to create a stable stand of native vegetation that is resistant to further invasion (D'Antonio & Chambers 2006; see Chapter 20). The degree of success of this undertaking depends upon the extent and density of the invasion, the extent to which soil and ecosystem characteristics have been altered, life histories of **invasive** and **native plant species** and interactions among these characteristics (Table 11.3). Invasive plant species include all life forms from herbs to shrubs and trees, and the range of techniques to control them includes mechanical removal (cutting,

mowing, fire and solarization), chemical control with herbicides, and biological control by insects or other herbivores. Some techniques that have been effective in MCRs are reviewed here.

Often, the extent of invasion is too large for mechanical and chemical control to be effective across large and complex landscapes. In these cases, efforts have focused on isolated populations or preserves that are particularly threatened by alien invasives, as in the case of the riparian tree and shrub invader *Tamarix ramosissima* in California. This tenacious species has been **eradicated** by hand control and herbicide in defined stands, but introduced *Tamarix* still dominates the riparian zone of thousands of kilometers of rivers in the western United States. Following careful testing, a host-specific biocontrol beetle, *Diorhabda elongata*, from Eurasia, has recently been released, and is now beginning to show good results in efforts to control *Tamarix* across large areas (Ehleringer *et al.* 2009). Where normal riparian hydrologic regimes still exist, native riparian trees are recolonizing and the abundance of the invasive *Tamarix* is declining.

Invasive species can be more readily controlled if they have not invaded at high density. In the Azores, that have woodlands of Mediterranean origin but an oceanic climate, low-density invasive trees were removed with cutting and herbicide from a laurel forest, followed by successful establishment of planted native trees (Heleno *et al.* 2010). The moist climate likely promoted establishment, as seedlings in a summer-dry Mediterranean climate may suffer higher mortality. Alien trees also re-established from seed in this experiment, but in a real restoration situation would need to be controlled by future treatment. Native

Table 11.3 Probability of restoration success after the invasion of alien species based on invasive species characteristics.

Invasive species characteristic	Probability of restoration success	
	Higher	**Lower**
Extent of invasion	Small	Large
Control technique	Biological control	Mechanical or chemical
Plant size	Large (woody)	Small (herbaceous)
Density of invasion	Sparse	Dense
Recolonization ability	Low	High
Seed bank longevity	Short-lived	Long-lived
Seed bank density	Sparse	Dense
Soil and ecosystem feedback	Minor	Major

insects and birds increased in abundance after control of invasive trees, an important finding given the conservation value of these remnant native forests (Heleno *et al.* 2010). The Asian invasive Tree of Heaven, *Ailanthus altissima*, was successfully controlled with tree cutting plus herbicide application in Spanish woodlands, but the authors did not report vegetation recovery (Constán-Nava *et al.* 2010). They recommended afforestation to avoid future recolonization of *A. altissima*. If invasive plants are controlled while their populations are sparse or their areas are small, any subsequent escapes, spreading or reinvasion can be controlled with relatively small efforts (D'Antonio & Chambers 2006).

Recolonization or re-establishment ability by **native species** is a critical prerequisite to restoration following invasive species control. In the study by Heleno *et al.* (2010), native seeds became the most important component of the seed rain just one year after clearing invasive trees, and the authors suggested this showed a renewed **successional trajectory** leading toward a native-dominated community. By contrast, Mediterranean annual grasses were the most abundant species to recolonize in California coastal sage scrub after mowing and grass-specific herbicide that reduced the alien grasses (E.B. Allen *et al.* 2005). Native shrubs had limited establishment under the constant impact of colonizing invasive species, and removal of invasive species was recommended every 5 years to maintain native dominance (E.B. Allen *et al.* 2005). Similarly, the Mediterranean grass barbed oats (*Avena barbata*) may compete with establishing native shrubs in abandoned farmland in Australia, and must also be periodically controlled (Standish *et al.* 2008). The Australian site had low soil nutrients (Standish *et al.* 2008), while the California site had high soil nutrients (E.B. Allen, *et al.* 2005), indicating that Mediterranean annual grasses have a broad ecological amplitude for invasion, especially barbed oats, which occurred in both study sites.

Seed bank density and longevity are also critical factors in the invasion process. The seed bank density of alien grasses was two orders of magnitude greater than that of native forbs and shrubs in California coastal sage scrub (Cox & Allen 2008). Fortunately, the seeds of many alien grasses – especially the dominant *Bromus madritensis* and *B. diandrus*, both from the Mediterranean region – are short lived, so that carefully timed spring fire (before the seeds have dispersed) has been a useful tool in restoring perennial grassland

(Gillespie & Allen 2004). However, given the ability of these annual grasses to recolonize or re-establish, fires must be set every 5–7 years. This high frequency of fire is a good tool for restoring perennial, fire-adapted grasslands, but is incompatible with native shrub or tree recovery, as frequent fires will not allow woody plants to establish or grow to maturity. Other means of alien annual-grass control are needed for shrub and woodland restoration, such as timed grazing in spring to reduce annual seed production. Seed bank longevity is also a factor in the successful control of *Tamarix*, which has a short-lived seed bank; the few seedlings that establish after mechanical or chemical control can be readily eliminated with follow-up treatment.

The degree of soil or ecosystem alteration following invasion will also determine the feasibility of restoring invaded systems in MCRs. Invasive *Acacia saligna*, from Australia, has caused elevated soil nitrogen of South African fynbos, which impedes restoration efforts because the native shrubland is adapted to low levels of soil nutrients (Yelenik *et al.* 2004). Hand clearing of the invasive shrub promoted colonization by *Ehrharta calycina*, a nitrophilous native perennial grass. As a result, it was recommended that fire or mulch be used to control high soil N and promote re-establishment of native shrubs. Bark mulch was effective in immobilizing N and allowing establishment of native shrubland that was invaded by alien Mediterranean annual grasses in California (Zink & Allen 1998).

As mentioned earlier, Mediterranean Basin woodlands and shrublands are apparently more resistant to biological invasions than those of other MCRs, and thus are – in theory – more amenable to restoration. Indeed, invasive plants from the Mediterranean Basin often come to dominate in other MCRs because they are more aggressive than the native species, or because environmental conditions (e.g. fire, grazing and soil nutrients) have changed so that the native species are no longer able to thrive. Some of the environmental drivers may be controlled (e.g. reducing fuel for fire or reducing grazing intensity), while others are less easily manipulated, such as reducing soil N on a large scale. Another approach to restoration is to select native species with physiological traits that inhibit invading species, and establish vegetation that is resistant to invasion (Funk *et al.* 2008). This approach will succeed if ecosystem disturbance rather than species traits have allowed invasion. However, if the environmental drivers have been controlled and the invasion is still ongoing, as has been observed in California coastal

sage scrub, then periodic mechanical or chemical control may be required (E.B. Allen *et al.* 2005), and invaded systems will require long-term management to achieve and then maintain desired conservation values.

11.2.5   Restoration of fire regimes

One of the challenges in landscape management of Mediterranean ecosystems is to restore, or reinstate, natural (historic) fire regimes. For this purpose, both fire suppression and prescribed fires may be appropriate tools to test. However, prescribed fires have also been used to reduce fuel loads and fire intensity without examining the natural fire regime. Indeed, restoring fire regimes is never a simple task because of the impact on and implications for native biodiversity. The main difficulties are as follows:

• Natural fire regimes vary spatially with climate and vegetation structure, and different ecosystems have species adapted to different fire regimes. Moreover, the natural or historic fire regimes are not always known. In ecosystems with surface fire regimes, fire scars from old trees can be used to infer past fire regimes; however, in crown-fire ecosystems (e.g. the chaparral- or matorral-like communities of all MCRs), this is not an option as most trees are fully burned and the main stem dies after the fire. In ecosystems intensively and perhaps over-used for millennia, including many in the Mediterranean Basin, old trees are very rare and can hardly be used for fire regime estimation; in such situations we lack an historical reference. Aerial photography and remote sensing have been used to reconstruct fire regimes to the earliest available imagery.

• Most fires in natural fire regimes occur in summer, when vegetation is most flammable, while for safety reasons most prescribed fires are set in the wet winter season. However, plants and animals exhibit different physiological and phenological conditions in different seasons, and as a result prescribed fires may not have the same effects and impacts as natural fires. For instance, birds nesting in spring are more susceptible to prescribed than to natural fire, and there is much evidence of varying plant regeneration responses to prescribed fires in differing seasons (e.g. Keeley 2006).

• Mediterranean climate and landscapes are very attractive to humans, and are among the most populated ecosystems on Earth, and the most visited by tourists. In most MCRs, people colonize landscapes in houses interspersed with vegetation, generating a large wildland–urban interface. In such conditions, natural fire regimes are difficult to restore without affecting human activities and infrastructures. There is still a lack of cultural, political and socio-economic understanding of living in flammable ecosystems (Pausas & Vallejo 2008).

• As mentioned above, invasive alien species are one of the major problems in MCRs and, with the exception of the Mediterranean Basin (where post-fire aliens are relatively uncommon), and frequent fires often increase the opportunities for invasion by alien plants (Keeley *et al.* 2005). Given the current high (unprecedented) propagule pressure of alien species, restoring natural fire regimes may imply the increase of invasive alien plants (Keeley 2006). This often leads to a challenge for managers who must choose between restoring natural fire regimes and altering those fire regimes to favour communities of native species, while altering the natural community structure.

In addition to these difficulties to restore the historic fire regime, climate is changing very fast in many or all of the MCRs, and due to the strong link between climate and fire, restoration of fire regimes should also be conducted with full consideration of projected climatic scenarios of coming decades. Moreover, climatic change also has consequences on other factors that may modify forest structure and flammability, such as drought and pests. According to some researchers (e.g. Fulé 2008), such considerations make local historical references of limited value while references from drier sites become more appropriate. Needless to say this is a fertile topic for debate, and long-term research and adaptive management will be necessary on a site-specific basis.

11.3   PERSPECTIVES

Major challenges for woodland and shrubland restoration in Mediterranean climate regions must be confronted in the coming decades. Some of these relate to previously identified, but still poorly implemented, needs that have emerged in the course of maturing restoration theory and practice, such as project evaluation and social involvement. Other research and development horizons relate to new and, above all, rapidly changing climate scenarios in the non-analogue **Anthropocene Era** we are now entering. For example, restoring sustainable fire regimes, while

also reducing wildfire threats to humans and built environments in fire-prone areas, is certainly of the most challenging issues in the land management and restoration of MCR woodlands and shrublands.

11.3.1   Evaluation and social involvement

Ecological restoration has a strong socio-economic component, particularly in highly populated areas such as the MCRs, where human societies have co-evolved with landscapes for centuries or millennia. Thus, social actors should participate in defining the objectives of restoration actions and through the evaluation process. The ecosystem service approach (Millennium Ecosystem Assessment (MA) 2005) provides an excellent opportunity to involve people and encourage participative management. There are many examples of such initiatives, and on the use of decision-making tools, such as multicriteria decision models, to support the identification of restoration targets in MCRs (Díaz-Balteiro & Romero 2008).

In order to improve our understanding of the success or failure of restoration actions, there is a need for long-term monitoring and evaluation of restoration actions. Evaluating ecological restoration success on the ecosystem and landscape scales can be performed using carefully selected suites of indicators (e.g. Aronson & Le Floc'h 1996; Vallauri *et al.* 2005). Although widely accepted standard protocols are not yet available, recent initiatives integrate biophysical and socio-economic aspects of restoration projects (e.g. Bautista & Alloza 2009, for Mediterranean Basin woodlands and shrublands).

11.3.2   Restoration and climate change

Woody vegetation that evolved in the Mediterranean Basin during the Pliocene shares common features such as sclerophylly, evergreeness and the ability to resprout after fire of many species (Blondel & Aronson 1999). However, these life-history traits are associated to Tertiary lineages that arose from tropical and temperate ancestors prior to the appearance of Mediterranean climate conditions in the last 3–5 million years (Herrera 1992; Blondel *et al.* 2010), and the Quaternary taxa evolved under Mediterranean climate in the various MCRs in the world do not show such convergent traits (e.g. they are not sclerephyllous, and many of them do not resprout after fire) (Verdú *et al.* 2003). Therefore, MCRs show a long-standing common background of characteristic woody taxa that have survived drastic climate change, both slow and fast. An open question is whether these species will endure projected climate change in their respective MCRs, in a human-altered and much more fragmented habitat.

Climate change must be taken into account when restoring ecosystems at the present time, considering that they should be functional by the middle of the twenty-first century under changed climate. It has become increasingly clear that knowledge of the past is not necessarily a good – or even sufficient – guide for understanding or planning for the future (Pahl-Wostl 2007). This is particularly so when considering the adaptability and acclimation capability of various species or functional types to a projected new climate, and the associated fire regime conditions. For example, no one knows how ecotypes and genotypes of Mediterranean plant species will respond to the projected intensification of drought and new, more severe fire regimes. Taking a precautionary principle, it is probably wise to fully explore the adaptation and acclimation potential of native species beyond local seed sources (Crowe & Parker 2008) before assisting the migration of currently alien species or subspecies from drier areas. To cope with the *uncertainty* induced by climate change, restoration and management must be **adaptive**, trying to improve ecosystem resistance and/or resilience, and managing landscapes to facilitate species migration (Stephens *et al.* 2010).

Increased knowledge of the pattern of climate variability may be used to plan restoration activities. In many Mediterranean semi-arid and arid regions, interannual variability in precipitation is strongly associated to El Niño Southern Oscillation (ENSO). Increased rainfall during ENSO events is crucial for plant recruitment and productivity in these systems and may cause long-lasting effects on vegetation depending on the prevalent herbivore pressure (Holmgren & Scheffer 2001). Indeed, field studies in arid zones of North America have indicated that successful recruitment of woody vegetation takes place during rainy ENSO events, especially when herbivores were excluded (Bowers 1997).

Holmgren and Scheffer (2001) proposed that increased tree and shrub establishment during rainy ENSO events could be improved by controlling main herbivores, thereby facilitating a switch of degraded

semi-arid ecosystems towards a more productive state. Because of positive feedbacks, a more productive state would tend to be maintained despite the relatively short duration of the rainy pulse that triggered the increase in primary productivity. This is an attractive proposal with major implications for future restoration programmes of original Mediterranean semi-arid and arid shrublands and woodlands, since ENSO events are potentially predictable several months in advance, they are connected to climate in distant MCRs, and model forecasts are steadily improving (Goddart *et al.* 2001). ENSO events have become more frequent in the last few decades, and their frequency might increase further as a result of global climate warming (Timmermann *et al.* 1999). We could therefore take advantage of these changes by coupling reforestation programmes and herbivory control to forecasted high rainfall ENSO events to enhance the probabilities of plant establishment at probably lower costs than those involved with traditional restoration programmes. Hopefully, increasing knowledge of projected future climate systems will allow optimizing restoration timing by forecasting, and taking advantage of, recruitment cycles.

MCR ecosystems have been subjected to long-term human pressure that has deeply modified their characteristics and spatial distribution, often resulting in degradation deserving restoration actions. Nowadays, new disturbances emerging since the post-industrial era interact with old disturbances, and on top of them global change is introducing further stress and complex changes in the disturbance regime. Biophysical approaches and technologies to address ecological restoration in MCRs have developed quickly during the last two decades, although mostly considering static references and targets. Under the scientifically acknowledged rapid dynamics of global change, the current challenge is to adapt ecological restoration theory and practice to these accelerated dynamics, including the socio-economic perspective. MCRs, with their extreme complexity, both biophysical and social, would be excellent pilot regions for testing new ecological restoration approaches in the face of global change.

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