Ecological restoration in mediterranean-type shrublands and woodlands.

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To be cited as:

Mucina, L., Bustamante-Sánchez, M.A., Duguy, B., Holmes, P., Keeler-Wolf, T., Armesto, J.J., Dobrowolski, M., Gaertner, M., Smith-Ramírez, C. & Vilagrosa, A. 2017. Ecological restoration in mediterranean-type shrublands and woodlands. In: Allison, S. & Murphy, S. (eds.), *Routledge Handbook of Ecological and Environmental Restoration*. Taylor & Francis, Abingdon, UK, pp. 173–196.

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ECOLOGICAL RESTORATION IN MEDITERRANEAN-TYPE SHRUBLANDS AND WOODLANDS

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Introduction

The Mediterranean-type ecosystems (further MTEs) are limited to five regions on Earth (Cowling *et al.* 1996): Mediterranean Basin, California, Central Chile, the Cape of South Africa and the Southwest (and partly South) Australia. These regions are characteristic of western ocean coastlines in warm-temperate latitudes characterized by descending water-deprived ethesial winds. They are invariably transitional between temperate forests and semi-deserts. Precipitation seasonality and prevalence of winter-rainfall/summer drought cycling are regular, although not exclusive to the MTEs (Blumler 2005; Rebelo *et al.* 2006). Fire has been part of the natural regeneration cycles and undoubtedly also evolutionary history of the scrublands and woodlands (perhaps except for the Chilean MTE) for millions of years. The Northern Hemisphere MTEs and the Central Chilean MTE are home to relatively young geologically and climatically dynamic landscapes. The MTEs of the African Cape and Australia are, on the other hand, geologically quiescent and climatically buffered – most of these regions qualify as Old Stable Landscapes (Hopper 2009; Mucina and Wardell-Johnson 2011).

MTEs are evolutionary hotbeds and musea: they are home to several global centres of biodiversity (Myers *et al.* 2000) and have about 20 per cent of total floristic diversity in an area covering just 5 per cent of the land surface. Vegetation of the MTEs is typically sclerophyllous shrublands, however (pine, oak, eucalyptus) woodlands are also important.

Besides the enormous biodiversity, the regions supporting MTEs have been under human pressure for a long time. Some (Mediterranean Basin and its eastern outposts in the Middle East) have been the cradle of agriculture and have seen the rise of many civilizations. Past and present human use put these ecosystems under pressure and where possible and feasible, restoration of these ecosystems emerged as one of the ways for their wise, future-oriented management. Each of the partial MTEs is exposed to multiple challenges of rehabilitation and a profound review of these is beyond the scope of this chapter. Therefore we have embarked on featuring the dominant rehabilitation focus in each MTE.

Mediterranean woodlands and shrublands

Challenges

Major degradation processes affecting the Mediterranean Basin (hereafter the Mediterranean) ecosystems are related to the long-term overuse of natural resources (overgrazing, woodland clearing, invasive alien species), increasing population pressure and associated political issues (such as urban sprawl and alteration of fire regimes) generating threats to sclerophyllous shrublands and woodlands in the entire Mediterranean. Paradoxically, land abandonment may play a negative role too. Steep moisture gradients spanning sub-humid and arid climate zones within the Basin, together with the natural environmental heterogeneity and the diversity of land-use histories underpin great ecosystem variability and hence, a wide range of degradation scenarios (Vallejo *et al.* 2012a). Desertification is affecting large areas in dry lands, reducing soil productivity (Vallejo *et al.* 2012c). In the past few decades, large intensive wildfires increased in frequency and intensity in the European Mediterranean landscapes, except in arid areas where fires are fuel-limited, imposing a serious threat both to natural ecosystems on one hand and to human life, property, and well-being on the other (Duguy *et al.* 2013; Moreno *et al.* 2013). Synergistic interactions between severe fires and ongoing degradation processes may be especially acute in vulnerable semi-arid ecosystems and trigger strong changes in ecosystem composition and structure (Moreno *et al.* 2013).

Climate change projections for the Mediterranean foresee increasing extreme temperatures and decreases in both rainfall and relative humidity (Kovats *et al.* 2014). These changes are predicted to foster longer fire seasons and more intense fires (Duguy *et al.* 2013). Under such conditions, post-fire regeneration will likely be impeded, hence diminishing resilience of plant communities (Delitti *et al.* 2005) and promoting opportunistic alien species (Lloret *et al.* 2003). Once the degradation thresholds have been crossed (e.g. by loss of keystone species), certain processes may reduce the ability of supporting spontaneous regeneration. Then, the vegetation, and ecosystems in general, may be restored only through human intervention in the form of restoration actions and manipulations (Vallejo *et al.* 2012a).

Some major challenges that ecological restoration in the Mediterranean is confronting include:

- definition of proper models of vegetation dynamics to use as a reference (e.g. natural, or 'historic', fire regimes), keeping in mind that in the light of global change local historical references may become of limited value while references from drier sites may become more appropriate (Fulé 2008);
- increasing need of long-term research and adaptive management leading to a better understanding of the abiotic and biotic factors acting as drivers of ecosystem functioning in order to identify processes hindering post-disturbance natural recovery and better selecting the areas and the ecosystem components or functions to be restored;
- promotion of better governance, based on broad social participation and diffusion networks in order to meet new demands on natural environment and spread best practices associated with land management (e.g. the promotion of prescribed fire, which entails a social, political and technical challenge, requires social acceptance and adequate regulation); and
- promotion of quality control and scientifically based evaluation of projects for optimizing restoration investments and delivering feedback from restoration experiences into the improvement of restoration processes.

Theoretical underpinning

Reforestation and afforestation have been the restoration actions traditionally implemented in the Mediterranean countries. Technically not considered ecological restoration projects as understood today, these actions addressed some of the broad aims of restoration, such as reduction of soil erosion and runoff, recovery of natural forests, and the like (Vallejo *et al*. 2012a). The 'tree-oriented' approach, based on Clementsian successional models and still applied by forest administrations, consists of (re)introducing one or several keystone species (generally pines), acting as 'ecosystem engineers' that are expected to modify the habitat and facilitate the establishment of late-successional species, thus fostering successional trajectories towards an ideal state, often identified as potential natural vegetation (Cortina *et al*. 2011).

This old paradigm recently shifted towards new approaches embracing successional state-and-transition models identifying likely trajectories and desired potential states on the basis of community composition and structure, ecosystem functioning and capacity to provide goods and services (Cortina *et al*. 2011). New projects generally consider a higher diversity of woody species, positive plant–plant interactions (i.e. facilitative effects), less aggressive plantation techniques, and smaller extent of the interventions.

Despite the high diversity of degradation and restoration needed in scenarios encountered across the Mediterranean, some objectives should be common to all restoration projects (Vallejo *et al*. 2012a), such as soil and water conservation, increase of ecosystem resilience to current and future disturbance regimes, promotion of native biodiversity while eradicating alien invasive species, and improvement of landscape quality and provision of ecosystem services.

Approaches to restoration

Under climate change projections, restoration strategies and techniques have to be adapted to increased drought stress and fire. In this sense, two types of interventions have probably received the most attention from ecological restoration research across the Mediterranean over the past few decades. These include *post-fire restoration* for mitigating or reversing negative fire impacts (often caused by novel combinations of fire regime and other disturbances in fire-dependent ecosystems), and *restoration of vegetation cover* in ecosystems affected by desertification and biodiversity loss in semi-arid areas.

Planning post-fire restoration requires an understanding of how the fire regime is affecting ecosystem fire resilience and the identification of the specific degradation processes triggered by fire (Vallejo and Alloza 2015). Restoration should address soil conservation in the short term (< 1 year) and the recovery of ecosystem integrity (function and structure, including biodiversity) together with ecosystem services in the longer term.

In ecosystems showing high erosion and runoff risk, with low short-term plant regeneration capacity, emergency rehabilitation actions are needed (Vallejo *et al*. 2012b). Two main (non-exclusive) soil protection techniques are used, such as *seeding* with fast-growing native species and, *mulching* with various kinds of organic materials.

In a second stage, vegetation recovery is the key factor for restoring soil productivity. Post-fire regeneration strategies of dominant species determine recovery rate (Keeley *et al*. 2012). Resprouting species allow a faster recovery of species composition and abundance than obligate seeders. The (re)introduction of native woody resprouters is thus recommended to increase fire resilience (Valdecantos *et al*. 2009).

The aspects to be considered for improving plantation success are plant species selection, nursery and planting techniques (Chirino *et al.* 2009; Duguy *et al.* 2013). Selection of species must be based on the natural flora and vegetation of the area and the specific biophysical characteristics of the site (Vallejo *et al.* 2012a). The number of species used in reforestation is increasing rapidly, moving from a reduced set of easy-to-grow species (mostly pines) to a large variety of native species (including shrubs).

Drought is the most critical factor hindering seedling survival across the Mediterranean (Vallejo *et al.* 2012c). Plantations have incorporated innovations for reducing seedlings' water stress (Chirino *et al.* 2009; Duguy *et al.* 2013; Vallejo *et al.* 2012c) by (1) increasing water-use efficiency (selection of drought-tolerant species and ecotypes, seedling preconditioning, improvement of below-ground performance and nutritional status), (2) increasing water supply (soil preparation and amendment for improving microsite conditions and resource availability), and (3) reducing water losses (tree shelters, mulching, microsite selection).

Some techniques may significantly enhance plant establishment in semi-arid ecosystems (Figure 13.1), particularly when combined in harsher sites (Kribeche *et al.* 2012; Valdecantos *et al.* 2014). Over the past decade, indeed, as advanced technologies were implemented, the relationship between seedling survival and drought length changed for experimental plantations in dry lands; survival rate doubled under a three-month drought (Vallejo *et al.* 2012a; Figure 13.2).

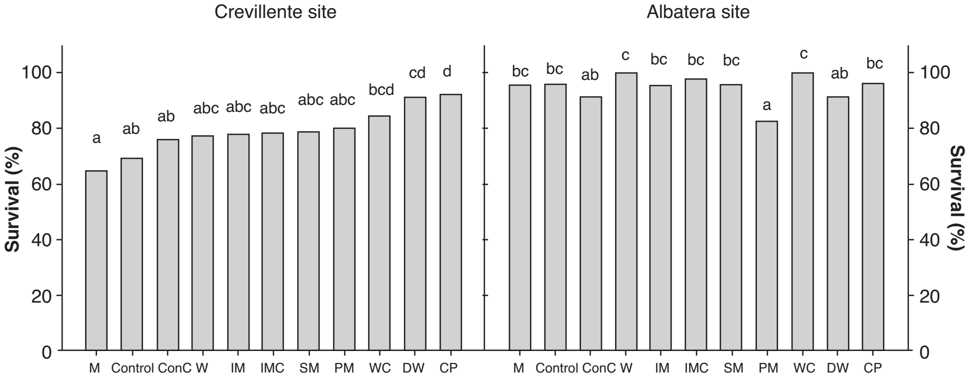

Figure 13.1 Survival of *Olea europaea* seedlings ten months after planting in relation to experimental treatments in two semi-arid stations (rainfall was 10 per cent higher in Albatera during the studied period). Control: traditional planting holes; M: Microcatchment; W: Deep water application (1.5L, twice); IM: M+waterproof surface upslope of holes; SM: M+stone mulch on soil surface; PM: M+plastic mulch on soil surface; DW: IM+stone mulch+2preferential water pathways; CP: 2.5L buried clay plot (filled twice); ControlC: Control+C; IMC: IM+C; WC: W+C, where +C is application of composted sewage sludge at an equivalent rate of 22.5Mg/ha. Different letters indicate significant differences by the log-linear analysis ($p < 0.05$)

Source: modified from Valdecantos *et al.* (2014)

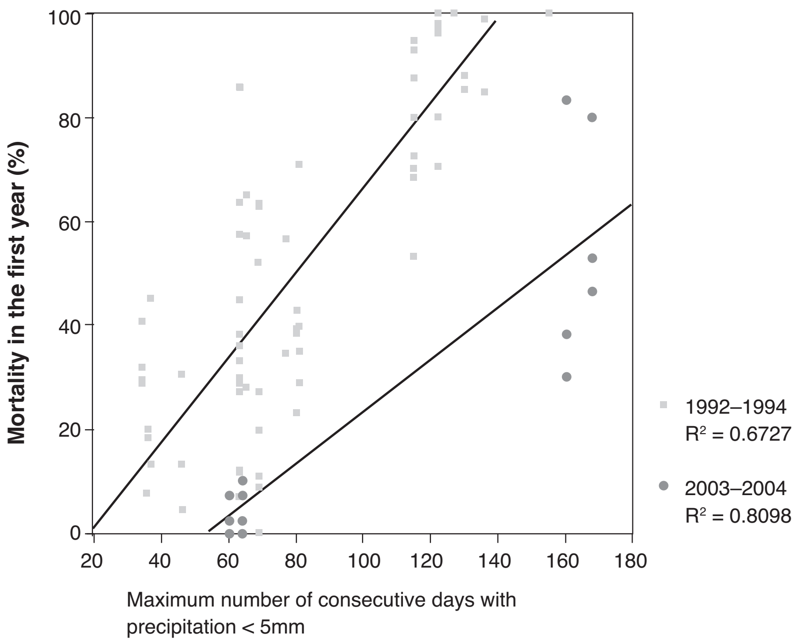

Figure 13.2 Seedling mortality in relation to the dry period length during the first post-plantation year, for several native species planted in eastern Spain. The first set of plantations (1992−1994) used conventional techniques at those times. The second set (2003−2009) used recent technical innovations

Source: modified from Vallejo *et al*. (2012a)

Californian coastal sage scrub and chaparral

Challenges to ecological restoration

Geographic set up and vegetation patterns

Mediterranean-type ecosystems occur in North America throughout the California Floristic Province (CFP) west of the Sierra Nevada and Cascade Mountains crest and the Transverse and Peninsular ranges from northwest Baja California (Mexico) to southwest Oregon.

California has the most extreme summer drought of all the Mediterranean-type climates (Vasey *et al*. 2014). Between May and September, there are almost no significant rain events. Its endemic shrublands have developed two divergent life history strategies which enable them to contend with this extreme summer drought: the shallow-rooted drought-deciduous approach, and the deep-rooted, evergreen sclerophyll strategy.

The major ecosystems of the CFP include California broadleaf and coniferous woodlands, California prairie (composed of annual and perennial grasses, graminoids and broad-leaved herbaceous species), and two shrubland types commonly known as coastal scrub and chaparral. The California *coastal scrub* is a drought deciduous shrubland formation dominated by short-lived shrubs with shallow, spreading root structures. Leaves tend to develop during the winter

Box 13.1 Restoration of drylands in Southeastern Spain: the combined role of site conditions and reforestation techniques

Southeastern Spain is one of the areas most affected by desertification in Europe. In the framework of the *Spanish Action Programme to Combat Desertification*, the Spanish Ministry of Environment and the Valencia Government Forest Service implemented in 2003 the Albatera restoration demonstration project, under scientific advice of CEAM, University of Alicante and CIDE-CSIC. Albatera site is a 25 ha catchment located in Alicante province. Land degradation was driven by the synergistic effect of past management and harsh environmental conditions, such as scarce (around 280 mm year^{-1}) and highly variable rainfall and erosion-prone soils.

The project aimed at putting into practice the best available restoration techniques for degraded semi-arid ecosystems, being an example of successful collaboration and technology transfer between the scientific community and stakeholders.

The main objectives of the project were (1) to repair ecosystem functioning by creating functional vegetation patches that contribute to the re-allocation of water, materials and nutrients, (2) to increase ecosystem diversity, stability and resilience, and (3) to prevent further surface and landscape degradation, soil erosion and off-site damage.

Based on functional characteristics, seven landscape units were identified and specific actions (species selection and restoration treatments) were designed for each. Eighteen native evergreen species (trees and shrubs) were selected. Plantation techniques aimed at maximizing water collection and conservation (micro-catchments, mulching), organic amendment (compost) and minimizing abiotic stress (tree-shelters). Although survival and growth rates of the introduced species were highly variable, six years after planting, survival was close to 50 per cent on south-facing slopes (where treatments were accumulated), against almost 10 per cent in the same area for past reforestations (Kribeche *et al.* 2012). Diversity and plant cover were higher (about 10 per cent) than in non-restored sites nearby. Some species had flowered and fruited, contributing to the recovery of the area. Soil loss decreased in all units over the monitoring period. Results show that suitable species selection and technological innovations improve reforestation outcomes, particularly in harsher sites. The cost-benefit analysis for best-technology actions yields a positive balance.

rainy season, and many are shed, or replaced, with smaller thicker leaves in the onset of summer drought (Keeley and Keeley 1984). The sclerophyllous evergreen shrubland known as California *chaparral* occurs on well-drained soils, often on steeper and rockier settings than coastal sage scrub. It consists of a variety of long-lived shrub species all of which tend to have deep roots and evergreen sclerophyllous leaves ranging from nanophyll to microphyll size. This vegetation is largely restricted to the CFP from SW Oregon, and south to northern Baja California, Mexico. California chaparral shrubs are divided into two main life-history strategies: (1) *seeders* tend to be relatively short-lived, storing seeds in a soil seed bank, and reproducing naturally through stand-replacing fire events, and (2) *resprouters* are long-lived (hundreds of years), and build up large carbohydrate stores in enlarged underground root structures (Keeley and Davis 2007).

Nature of disturbance sources

Natural fire frequencies in the shrublands of the CFP are not particularly high, since lightning is limited to the occasional summer monsoonal thunderstorm emanating from Southern

Mexico. Natural fire frequencies for many sage scrub and chaparral landscapes have been projected to be 20–50 years and fire occurred primarily in the late summer and fall. Recurring high-frequency fires can deplete the seed banks of many chaparral and sage scrub species, especially those that regenerate largely from seed.

Resprouting chaparral species (Keeley and Davis 2007) are able to survive periodic fires, but can persist for decades without them. A dense cover generally develops in the first decade after a fire, and these shrubs dominate within 30 to 40 years as they overtop shorter-lived or shorter stature species. Long intervals between fires and mesic conditions promote the development of mesic resprouting chaparral species.

The California shrublands are coincident with several of the largest urban centres in coastal California including the Los Angeles Basin, San Diego and the San Francisco Bay Area. An estimated 80–85 per cent of the pre-European extent of sage scrub has been eliminated from its southern California range (Reid and Murphy 1995), replaced with anthropogenic landscapes of ruderal or seral herbaceous vegetation, in some cases driven by increased atmospheric nitrogen deposition through automobile and industrial exhaust (Allen *et al.* 1998).

Motivation for ecological restoration

Many wildlands in California have been eliminated by rapid urban growth over the past 50–60 years. Increasing awareness of natural values of wildlands have motivated urban inhabitants to seek more sustainable solutions to urban growth compatible with maintaining natural populations of plants and animals, and natural processes in these landscapes. Certain subtypes of sage scrub and chaparral are recognized as rapidly decreasing habitat for endangered vertebrates.

Theoretical underpinning

Both vegetation formations tend to respond to different seed germination agents. For example, sage scrub species such as *Salvia* spp. respond to smoke-borne germination agents while chaparral seeds tend to respond to high heat (Keeley and Fotheringham 2001). Long intervals between fires are needed to develop mesic chaparral stands due to their short-lived shade tolerant seeds and resprouting response.

Sage scrub restoration is generally less complex than chaparral restoration. Many coastal sage scrub species are considered to be highly opportunistic in their germination requirements (*ibid.*). Several sage scrub species seeds have simple smoke-induced germination, but also produce some seedlings without fire. Coastal sage scrub species are also wind dispersed, and many are unpalatable to herbivores.

Compared to sage scrub, chaparral restoration efforts have more difficulty with establishment due to more specific relationships with mycorrhizal fungi (Horton *et al.* 1999), larger, fewer, seeds with more specific dispersal strategies, lower recruitment, and longer establishment times due to palatable higher nutrition foliage to herbivory and effects of drought.

However, like chaparral, coastal sage scrub is vulnerable to type-conversion. Abundant and ubiquitous introduced non-native annual grasses and herbs produce flashy fuels and when ignited create a continuous herbaceous fuel bed. Frequent fires are perpetuated by proximity to human ignition sources and atmospheric nitrogen deposition encouraging rapid re-growth and thatch production. The mycorrhizal root associations of coastal sage scrub can be disrupted by clearing the native shrub cover in conjunction with ploughing or 'deep ripping', also precipitating a type conversion to alien-dominated vegetation (Bozzolo and Lipson 2013).

Approaches to restoration

State-mandated regional conservation planning of the 1990s involved a significant portion of the natural range of coastal sage scrub. In southern California, many local government agencies have established objectives, standards and success criteria for coastal sage scrub restoration plans. These typically include the following principles:

1. Restore and raise the ecological condition of a disturbed site to a high-quality condition, equal to the pre-disturbance condition.
2. Use locally adapted plant material to establish a self-sustaining habitat with appropriate plant species richness, diversity and composition, based on the original vegetation, the physical characteristics of the site, the biological context of the site, and the nature and degree of disturbance to the original vegetation.
3. The restoration plan should clearly state the desired result of the habitat restoration for the particular site and set forth standards and success criteria. The condition of the restoration site has a strong bearing on the method of approach. Standard approaches have included conventional horticultural practices, such as irrigation, fertilizer, cages and soil amendments.

Applicability of restoration approaches to California scrub vary widely due to the climate, topography and existing site conditions of the project. Restoration settings include a wide range of possible situations from lightly disturbed (e.g. removal of original scrub and replacement by unwanted non-native cover), moderately disturbed (e.g. soil profile disturbed by cultivation but original topography intact), or heavily disturbed (e.g. all original vegetation removed, soil removed, topography modified). Depending upon the circumstances the following restoration practices are generally considered necessary to ensure a reasonable level of success.

Box 13.2 Leona Quarry restoration, Oakland, CA

A rock quarry since the early 1900s, this roughly 53 ha area has been reclaimed as partial mitigation for a housing project occupying the lower portion of the site in 2005. The upper two-thirds of the site was re-engineered to reduce geological impacts based on the location along the geologically active Hayward Fault, and to prepare the site for a chaparral restoration project. An approved county conservation measure established to reduce urban sprawl in the eastern portion of Alameda County sent developers scouting for available sites within the existing metropolitan zone. The project in the city of Oakland gained credit through a conservation easement for increasing habitat for the endangered Alameda Whipsnake, by reclaiming chaparral and sage scrub habitat lost during the expansion of the quarry. Following slope stabilization, slopes were blanketed with strips of coconut fibre and straw. The exposed face of the quarry was then covered with a layer of topsoil amended with compost. High-pressure hoses were used to spray a hydroseed mixture of native grass seeds, paper mulch and a glue agent. About 2000 holes were augered and filled with soil for the planting of coast live oak (*Quercus agrifolia*) and coastal sage scrub and chaparral species largely local to the adjacent natural area (Figure 13.3), including *Salvia mellifera, Eriogonum fasciculatum, Lotus scoparius*, and chaparral species such as *Adenostoma fasciculatum* and *Frangula californica*. Planting was arranged in a regular pattern that alternated chaparral and coastal scrub species, following hydroseeding of grassland species across the stabilized slope. The area was irrigated and maintained for

Figure 13.3 Landscape of the adjacent Chimes Creek (California) watershed used as the restoration model and local species source area with matrix of coastal scrub, chaparral and oak woodland. Newly prepared slope in upper right background May 2007, about one year post-planting. White-flowered shrubs in foreground are the regionally dominant chaparral shrub *Adenostoma fasciculatum*; small rounded trees are *Quercus agrifolia*

three years using a network of drip irrigation tubes. A five-year monitoring period was established to ensure native plant establishment and invasive species removal.

After planting species cover increased slowly for the first five years. During that time wire mesh protective cones were established around the slower-growing chaparral species (*A. fasciculatum*, *F. californica*). Following a major rainfall year in 2010 the cover increased appreciably (Figure 13.4). However, most of the cover came from increases in coastal scrub species and early seral species such as *Baccharis pilularis*, which seeded naturally from adjacent off-site areas.

Sage scrub species grew quickly from seed and comprised the majority of cover ten years after restoration. Invasive exotics have colonized the site particularly on the west side adjacent to a local off-site source of alien *Genista monspellieanus* and *Cortaderia jubata*.

Ten years post-site preparation, *Adenostoma fasciculatum*, the local chaparral dominant, established only in protected cages due to heavy preferential browse by native mule deer, while short-lived *Baccharis pilularis* (right) grew fast and did not require assistance establishing.

Vegetation was at a maximum in 2011 following a high rainfall season, but has actually declined since, due to four successive low rainfall years, and continued browsing by local mule deer. Based upon the first decade following restoration at this site, approximation of native chaparral structure dominated by *Adenostoma fasciculatum* will take many more years to achieve.

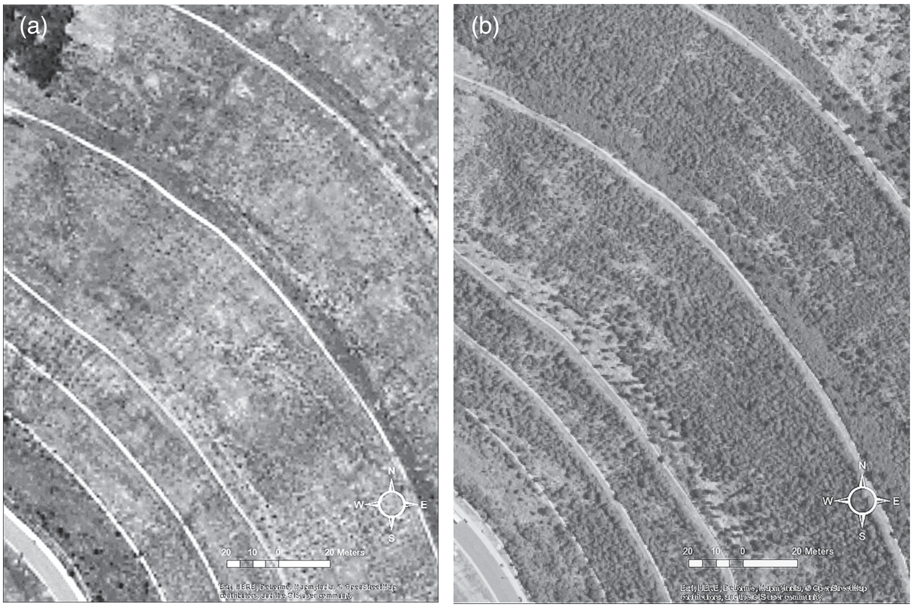

Figure 13.4 (a) August 2010 view of upper quarry five years following restoration, showing sparsely populated regular rows of shrub plantings, grass and herbaceous cover. Shrub cover is both native, naturally established *Baccharis pilularis* and potted and planted starts of *Adenostoma fasciculatum*. Thickly vegetated upper left corner is invasive exotic *Genista monspessulana*. (b) Shrub cover has increased as of this photo in July 2015. However, most is composed of naturally colonizing early seral species *Lotus scoparius* and *Baccharis pilularis*, not planted at the site. Of the planted sage scrub species, the highest cover was *Eriogonum fasciculatum*, followed by *Salvia mellifera* and *Artemisia californica*. The chaparral species comprised less than 1 per cent of total cover

Chilean sclerophyllous woodlands and matorral

Challenges

Anthropogenic disturbances have shaped the sclerophyllous forests and scrublands of Central Chile for almost five centuries. Such disturbances have been associated primarily with extensive use of firewood, cattle grazing, and more recently, agricultural and urban expansion, together with continuous degradation of vegetation by persistent burning. Fire, used by humans to clear vegetation for several centuries, greatly altered the historical disturbance regime in the Mediterranean-ecosystem region of Chile (Armesto *et al*. 2009).

The increased frequency of fire and logging in Central Chile has rejuvenated and changed the extent of remaining forests and scrubland, generating a predominance of areas dominated by shrubs such as *Acacia caven*, young forest stands (>50 years old) and a mosaic of small-size patches (<100 ha) (Schulz *et al*. 2010; Van de Wouw *et al*. 2011). Some native plant and animal species have declined to levels of quasi-extinction locally (e.g. *Beilschmedia berteroana*, *Beilschmedia miersii*, *Gomortega keule*; Hechenleitner *et al*. 2005). Efforts to restore community composition are faced with the problem that the scarcity and heterogeneity of native sclerophyllous forest remnants makes it difficult to define the reference systems to be used as targets for restoration. Knowledge of structure, composition, and environmental conditions of ancient remnant forest is typically unknown, but this knowledge is of great relevance for both restoration and rehabilitation planning.

A conceptual framework to guide restoration

Grounded in a conceptual framework based on knowledge of succession, disturbance, ecological filters and community assemblage rules (Temperton *et al*. 2004), we identify the factors that limit or facilitate spontaneous colonization in degraded areas of Mediterranean-type forest ecosystems in Chile and that may be applied to assist the regeneration of degraded areas. Based on the review of the existing scientific knowledge about forest ecosystems of Central Chile, we have put forward five general strategies to guide future restoration programmes in the region.

First, 'passive restoration' or assisted natural regeneration may be effective under certain circumstances. It should be favoured in degraded areas that are located near forest remnants that could act as propagule sources, or in south- and west-facing slopes, and other relatively humid sites in Central Chile (e.g. coastal areas, wet ravines). Such sites show lower seedling mortality and can recover from disturbance without active intervention. Depending on the specific site, sometimes it will be necessary to exclude herbivores (installing a fence). However, native vegetation could often recover even in the presence of herbivores (Fuentes-Castillo *et al*. 2012; Holmgren *et al*. 2000).

Second, the main limitations for successful establishment of native woody species in a degraded or open Mediterranean-type ecosystem derive from the combination of seasonal water stress, and the intensity of chronic disturbances such as fire and herbivory (Fuentes *et al*. 1983, 1984). Consequently, restoration strategies must be able to identify techniques to allow woody species to persist in the face of chronic stresses, which are part of the current environment. It will be advisable, therefore, to identify assemblages of species with complementary functional traits that may tolerate different kinds of stress and flexible characters that allow them to survive in open, degraded land (Laughlin 2014). For instance, if restoration is limited by high water stress, attributes such as hydraulic architecture, rooting depth or LAI could be related to differences in growth and survival. Traits that favour persistence should be selected and promoted within the species used in restoration.

Third, seed dispersal into open areas is often a limiting factor for succession and establishment. Because *dispersal vectors* are very important in the Mediterranean-type ecosystem of Central Chile (Armesto *et al.* 1987; Reid and Armesto 2011), succession can be initiated by planting species with fleshy fruits such as *Maytenus boaria*, *Lithraea caustica* or *Aristotelia chilensis*. These species will attract frugivorous birds to these areas to feed on their fruits, thus facilitating the arrival of fleshy propagules of other species and enhancing a diversity of colonizing species.

Fourth, *nurse effects* are an important plant–plant interaction facilitating plant establishment in stressful environments. It has been shown that the survival of woody seedlings is enhanced under the cover of pioneer shrubs (Becerra *et al*. 2011; Fuentes *et al*. 1984; Ovalle *et al*. 1999). In the case of sites without a shrubby cover, it may be advisable to actively promote the establishment of nurse shrubs.

Fifth, given the fact that *plant–bird interactions* often enhance the arrival of seeds of trees or shrubs into disturbed sites, and that nurse effects facilitate woody plant establishment, we propose that restoration programmes should follow a model of succession known as *nucleation* (Figure 13.5). In fact, this process spontaneously occurs in several places in the region (Fuentes

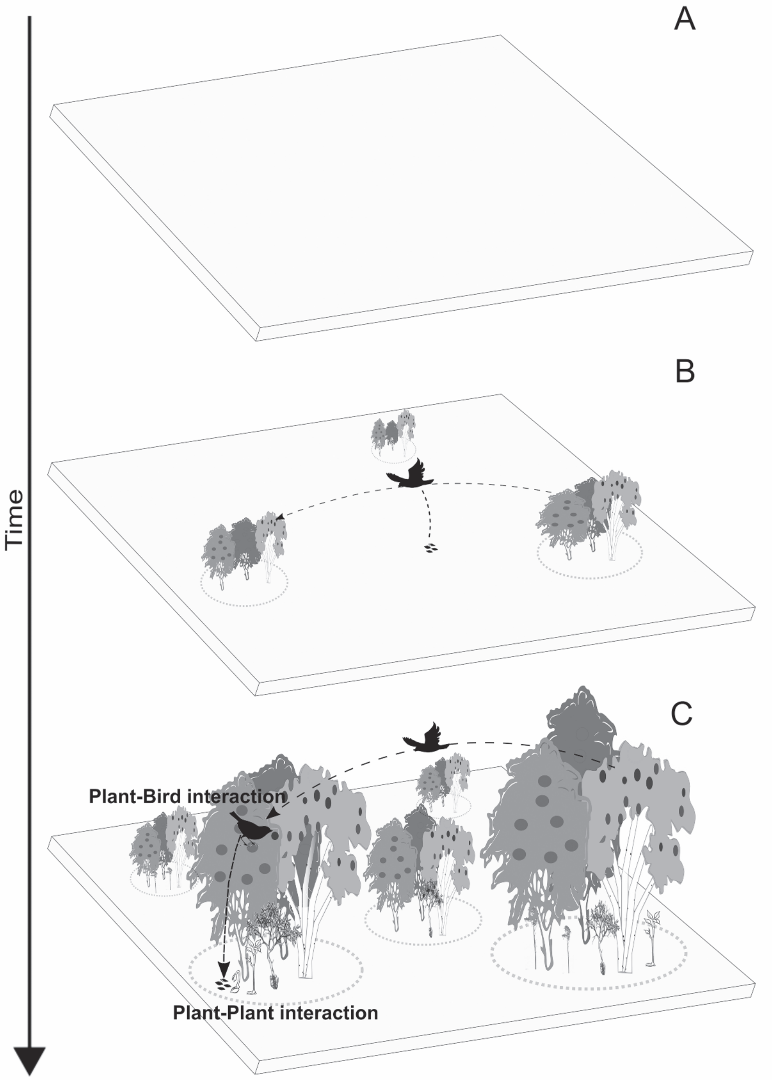

Figure 13.5 Graphic model of the nucleation process based on 'islands' or patches of planted trees. An area deprived from vegetation (A) is planted with blocks of multiple woody fleshy species (B). Once trees and shrubs have grown to sexual maturity (C), patches will expand and disperse seeds into neighbouring open areas. Patches will eventually coalesce (D) to produce a continuous canopy

Source: modified from Benayas *et al.* (2008)

et al. 1984; Fuentes-Castillo *et al.* 2012). Under this view, the planting of woody pioneers in open areas should be done in clumps, or artificial 'islands' (Benayas *et al.* 2008), including species with fleshy fruits. In the long term, these initial nuclei could promote natural regeneration over large spatial scales.

Box 13.3 Partnerships between scientists and practitioners are effective means for success

Experiments of ecosystem restoration are commonly conducted by researchers working in the science of restoration ecology. Frequently, researchers conduct quantitative assessments of the initial and long-term success of their experiments, in contrast to similar experiments performed by practitioners, which are assessed mainly through qualitative descriptions.

We discuss two case studies in Central Chile, from the point of view of practitioners and scientists, to show that working in partnership and sharing data between these two groups is important for the success of future projects:

1 The ecological reserve Oasis de la Campana (www.reserva.cl/proyectos_quienes.html), located in the Valparaiso Region, aims to create a space for the coexistence of humans and nature by conserving the natural vegetation and actively contributing to the recovery of human-degraded areas. To this end, they have worked on the recovery of 140 ha of degraded areas by hand-sowing *Acacia caven* seeds. Five years later, the open area has been filled by a dense cover of this shrub, regardless of the impact of herbivores (P. Moreno, unpublished report).
2 Experimental assays conducted by scientists in Valparaiso and the Metropolitan Region evaluated the effect of herbivore exclusion on woody regeneration. Fenced and unfenced plots were located in open pastures (covered by some individuals of *A. caven*). A higher proportion of woody seedlings became established under shrubs of *A. caven* in fenced plots than in unfenced plots (Miranda *et al.*, unpublished data).

From both examples, we conclude that it is clearly possible to increase the woody cover of areas devoid of vegetation by using low-cost techniques (hand sowing) to start up the recovery process. Once a shrub cover has been established, fencing may be necessary to enhance survival of established native trees. Therefore, combining results from both experiences has a greater value than each result taken separately.

Sclerophyllous shrublands of the Cape Region of South Africa

Introduction

The Cape Floristic Region (CFR) global biodiversity hotspot is located in the southwestern corner of South Africa and comprises only 4 per cent of the country. Mediterranean-type ecosystems are confined to western regions of the CFR while the eastern areas also receive a portion of summer rainfall. The major ecosystems of the CFR are fynbos and renosterveld sclerophyllous shrublands. Fynbos occurs on nutrient-poor substrate, primarily sandy soils derived from quartzite, sandstone, granite and rarely shale and limestone, whereas renosterveld shrublands occur on more nutrient-rich soils derived mainly from shale and granite parent materials under a drier rainfall regime (Rebelo *et al*. 2006). Strandveld shrublands that occur along the coast on alkaline soils, Cape Thicket and Afrotemperate forest confined to fire-protected habitats, are smaller vegetation types in the CFR and shall not be dealt with here.

In fynbos shrublands, fire and water availability are the major ecological drivers of recruitment and community structure, whereas in renosterveld shrublands mega-herbivore grazing also appears as a key driver (Radloff *et al*. 2013). Both fynbos and renosterveld shrublands require restoration strategies tailored to the nature of major drivers and their (re)assembly. Here we focus on alien-invaded fynbos shrublands as an example of restoration.

Ecological degradation and restoration

Nature of disturbance

Fynbos ecosystems are highly threatened, primarily as a result of direct habitat loss for agriculture, urban development and mining. The second largest threat is invasions by alien plants. Several alien tree species, including pines, hakeas and acacias, have invaded extensive areas and have out-competed the fynbos to form dense stands that alter ecosystem function, including changes to fire regimes (Wilson *et al*. 2014). Further ecosystem degradation is caused by inappropriate management such as too-frequent fires, over-harvesting and over-grazing (Holmes and Richardson 1999).

Motivation for ecological restoration

Ecological restoration in fynbos is motivated for two main reasons: first, to restore ecosystem function and thereby improve ecosystem services, such as water production from mountain catchments that are invaded by alien trees (van Wilgen *et al*. 2012); and second, to improve both function and community composition in support of biodiversity conservation in degraded areas of biodiversity networks (Rebelo *et al*. 2011). Owing to the high number of IUCN threatened plant species in the CFR (Raimondo *et al*. 2009), species restoration also is an important goal, particularly in the highly transformed lowlands.

Theoretical underpinning

Conceptual framework

A conceptual framework for restoration developed by Holmes and Richardson (1999) derived protocols from fynbos recruitment dynamics, community structure and ecosystem function. Predicted outcomes based on this framework were upheld in restoration studies in upland fynbos, following degradation by dense alien vegetation (Holmes *et al*. 2000) as well as topsoil disturbance that simulated mining (Holmes 2001).

In terms of recruitment dynamics, the majority of fynbos species have persistent soil-stored seeds that are stimulated to germinate by direct and indirect fire-related cues. Ephemeral geophytes persist as bulbs, corms and tubers. In upland fynbos, such species may survive two fire-cycles of dense alien invasion as dormant propagules, enabling autogenic restoration following alien clearance and fire. However, perennials that rely mainly on sprouting (e.g. shrubs with lignotubers) and shrubs with canopy-stored seeds become locally scarce or extinct, respectively, following dense alien invasion. In order to improve community structure, active restoration of such under-represented guilds is required.

Fynbos community re-assembly follows an auto-succession, whereby all components recruit after fire and are represented in the aboveground community for varying periods according to their respective lifespans (Bond and van Wilgen 1996). Therefore it is appropriate in active restoration to re-introduce all components together, in the immediate post-fire (or exposed soil) stage. Owing to high fynbos diversity, particularly of beta and gamma diversity (i.e. high turn-over along environmental and geographical gradients), practitioners should be careful to collect and re-introduce appropriately adapted, local taxa.

Many fynbos species are structurally and functionally similar. From an ecosystem function perspective, such analogues could be considered redundant, yet are thought to improve resilience to perturbations. From a restoration perspective, it is essential to re-instate a balance of the most important functional guilds, including species representing the main growth forms, regeneration and nutrient acquisition modes. Wherever possible, several species in each of these guilds should be introduced in order to improve resilience in the restored community.

Restoration thresholds

Research in lowland fynbos indicated that not all the protocols discussed previously were universally applicable, as regime shifts would occur more rapidly in low-altitude, particularly sandplain ecosystems. A restoration threshold model has been applied to conceptualize the different responses to degradation (Gaertner *et al.* 2012a). A threshold is recognized as the point at which the dominance of regulating feedbacks that maintain resilience switch to a dominance of positive feedbacks that lead to loss of resilience. Seed banks of perennial fynbos species are shorter-lived in the lowlands, possibly owing to intensive small mammal activity, limiting the potential for autogenic restoration (Holmes 2002). In lowland fynbos a biotic threshold may be passed following only one cycle of dense aliens, compared to two cycles in upland fynbos. The dominant invasive alien in the lowlands (Australian acacias) alters soil nutrient cycling processes (Gaertner *et al.* 2011), with nitrogen enrichment causing positive feedback and regime shift to a weedy, herbaceous community (Yelenik *et al.* 2004). In this case, a second, abiotic threshold has been passed and restoration interventions also must address the altered soil chemistry.

After more extreme degradation, such as in previously farmed land devoid of native propagules and with altered soil conditions, restoring community structure will be a more challenging goal. In some contexts it may be appropriate to modify the goal to one of restoring a particular ecosystem function.

Approaches to restoration

The theoretical frameworks outlined previously are useful in restoration planning (Figure 13.6). The first step is to identify an appropriate restoration goal. This will depend on several factors, such as extent of degradation, conservation importance, future land use and available budget. Restoring composition as well as ecological structure and function generally will be the most exacting goal requiring the highest resource inputs. Topsoil should be stripped ahead of mining operations and conserved, then replaced during restoration as it contains the soil-stored seed bank and microbial symbionts that greatly enhance restoration outcomes. Where topsoil has been lost or modified, the return of a small amount of topsoil, together with appropriate soil amelioration, can help to overcome the abiotic threshold to restoration (Holmes 2001).

Where soil-stored seed banks have been lost (e.g. ploughed fields) or depleted after long invasion by alien vegetation, a biotic threshold has been crossed and active restoration in the form of sowing and/or planting major fynbos structural components will be required to meet the goal of restoring ecological structure and function. This intervention should align with the natural post-fire recruitment regime in the autumn season, using fire-related stimuli to cue germination in sown seed. Where indigenous seed banks survive following dense alien

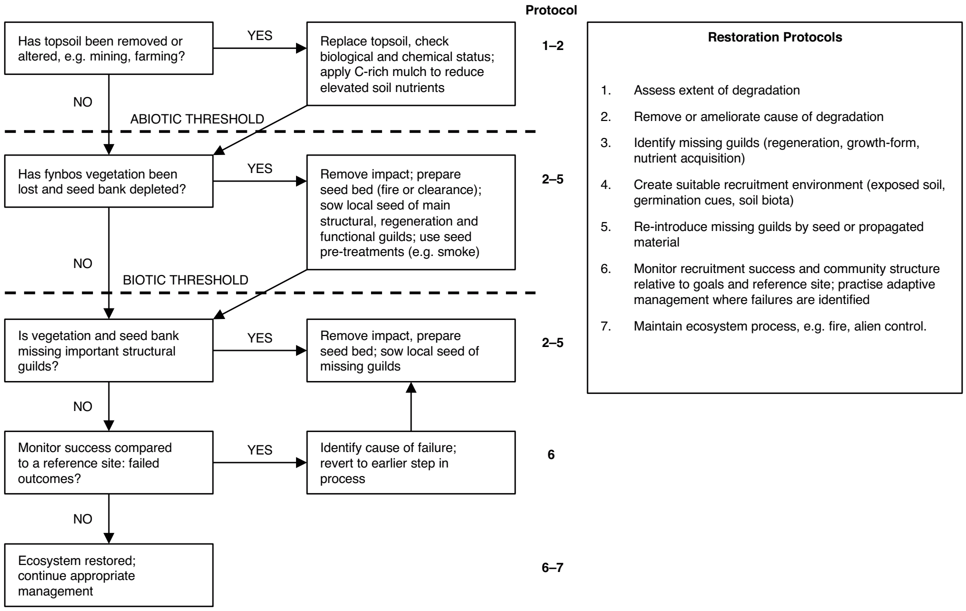

Figure 13.6 Flow diagram illustrating the degree of vegetation degradation and the minimum restoration actions required in order to restore fynbos community structure

invasion, the main intervention will be to re-introduce missing guilds, such as the overstorey proteoids with canopy-stored seeds. Ongoing follow-up control to ensure that invasive alien species do not re-establish will be essential. Restoration success will vary among sites and year of implementation, therefore it is important to set specific targets and timeframes that can be monitored towards meeting the restoration goal. This will assist in deciding when additional interventions may be required. A case study of restoration of Cape shrublands is outlined in Box 13.4.

Western Australian Mediterranean shrublands and woodlands

Ecological and evolutionary background

Within the structurally (and climatically) typical Mediterranean-type ecosystems of Western Australia (and smaller regions in South Australia, Western Victoria and New South Wales), the most iconic are the kwongan shrublands (incl. *Banksia* woodlands) on deep leached sandy regolith. Closed Jarrah and Wandoo woodlands on lateritic and shallow granite soils are yet another iconic vegetation type falling within the category of the MTE ecosystems. Technically, the latter ecosystem would qualify as dry forest, yet it has a number of features in common with the kwongan.

Besides the climatic characteristics common to all MTEs, especially the seasonal drought associated with high evapotranspiration and enhanced by frequent desiccating winds originating from the neighbouring semi-deserts and deserts, the major ecological challenge to restoration is posed by the soils supporting kwongan and woodlands; these are the oldest, and probably the nutrient-poorest soils on this planet (Lambers 2014). Fire-dependency of the

Box 13.4 Ecological restoration of old fields at Flower Valley, near Gansbaai, South Africa

Ecological and financial feasibility of active restoration was studied on three different invaded sandstone fynbos sites on Flower Valley Farm in the CFR with the aim to identify cost-effective ways of restoring functional native ecosystems following alien plant invasion (Gaertner *et al*. 2011; Figure 13.7). We tested mechanical clearing, burning, different soil restoration techniques and sowing of

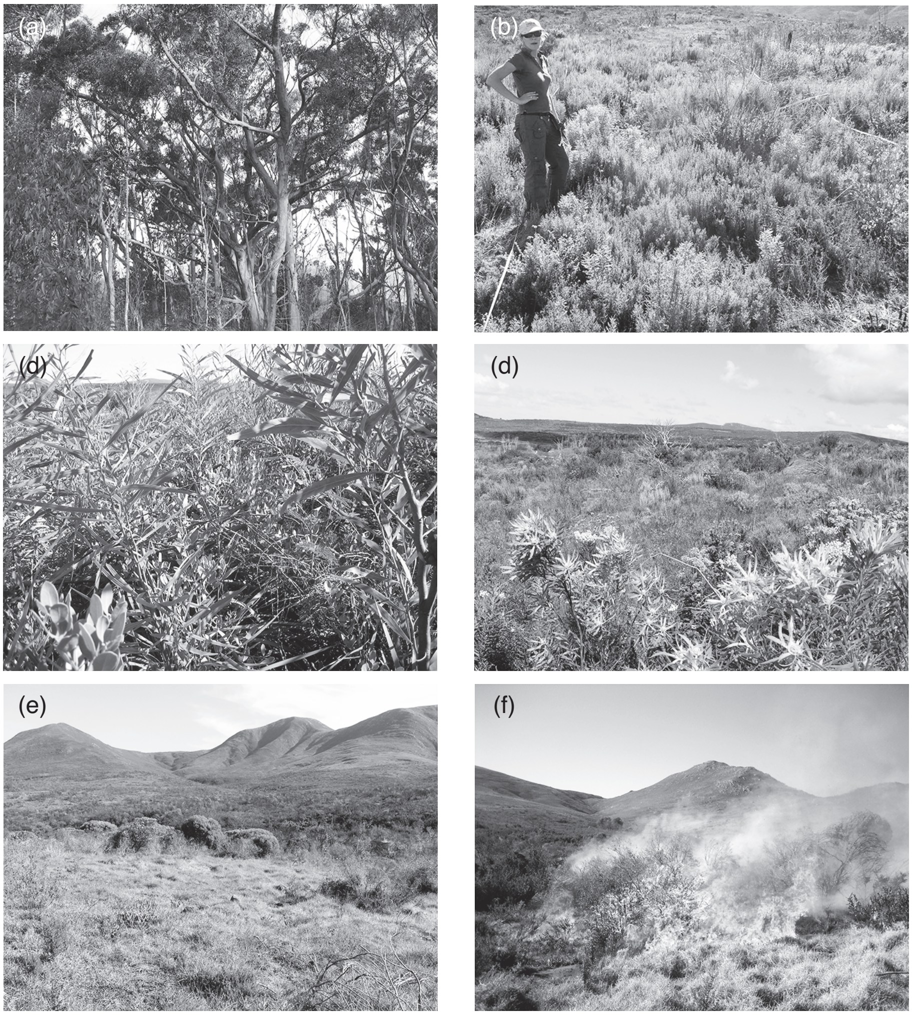

Figure 13.7 Restoration study sites at Flower Valley Farm, Gansbaai, South Africa. (a) Eucalyptus plantation (*Eucalyptus conferruminata*, *E. cladocalyx* and *E. gomphocephala*). (b) Seedlings coming up two years after the restoration. (c) Acacia thicket (*Acacia cyclops*, *A. longifolia*, *A. mearnsii* and *A. saligna*). (d) Reference site with mountain fynbos on acidic Table Mountain sandstone. (e) *Pennisetum clandestinum* (kikuyu grass) field subject to experimental fire (f)

Source: photos by Brummer Olivier

native species. We also investigated the possibility of creating incentives for private landowners by introducing fynbos species for sustainable flower harvesting. Restoration was successful: diversity and evenness of native plant species increased significantly at all three sites, whereas cover of alien plants decreased. However, sowing of fynbos species had no significant effect on native cover, species richness, diversity or evenness in the recently invaded *Acacia* thicket (*Acacia cyclops*, *A. longifolia*, *A. mearnsii* and *A. saligna*) and a formerly ploughed kikuyu (*Pennisetum clandestinum*) grass field, implying that, after one cycle of invasion, the ecosystem was still sufficiently resilient to allow autogenic recovery. But introduction of native species improved ecosystem structure, particularly major fynbos growth-forms such as proteoids and ericoids shrubs that otherwise would have been under-represented.

Income from flower harvesting following active restoration consistently outweighed income following passive restoration, but the associated increase in income did not fully compensate the higher costs. In conclusion active restoration can be effective and financially more viable than passive restoration, depending on the invasion characteristics (Gaertner *et al.* 2011, 2012b).

Australian MTEs probably has the longest evolutionary history, and therefore changes to the fire regimes (such as protection of the rehabilitation sites from naturally occurring fires) might create a new problem for the rehabilitation.

The kwongan scrub and eucalyptus woodlands on nutrient-poor soils are extremely species rich (Mucina *et al.* 2014), home to old lineages, staggering number of endemic, rare and endangered plant species. They are a national biodiversity treasure deserving World Heritage status. Due to poor soils, agriculture has not impacted much on kwongan (except for some scrub types such as woodjil). Small areas of kwongan are under active mining targeting titanium-rich minerals (Eneabba, Cooljarloo) and some Jarrah forests have been targeted because of high-quality bauxite (e.g. Gardner 2001; Koch 2007).

Regional rehabilitation challenges

Post-mining rehabilitation in water-deprived Western Australia is naturally water constrained. Despite most of the flora of the kwongan shrublands and the eucalyptus woodlands being adapted to semi-arid conditions, the initial stages of the rehabilitation (spread of top-soil, seed broadcasting, planting) require considerable amounts of initial water input that might not be available especially in summer time. Low nutrients naturally do not pose a major constraint on the rehabilitation since most of the flora involved is well adapted (or well-exapted) to the low nutrient levels. These plants also sport a plethora of nutrient-acquisition strategies (Lambers *et al.* 2008). However, many of those involve special biotic interactions (mycorrhiza, rhizobia) and failure to restore the microbial life in soil and the microbial-plant interactions lead to failure in restoring populations of so called 'recalcitrant', often iconic, species groups (orchids, sedges and the like). There is not much knowledge on the role of fire in rehabilitations (but see for instance Roche *et al.* 1997), but scanty observations support an idea that controlled application of burning in progressive stages of the rehabilitation might be profitable to restore populations of some recalcitrant seeders.

Theoretical underpinning

Successful, scientifically based post-mining rehabilitation is rooted in understanding of (1) the nature of vegetation dynamical pathways of the impacted vegetation, (2) the processes of plant community assembly, (3) the interaction of the restored biotic community with its environment, (4) the technological tools used in the rehabilitation, and finally (5) skill in application of the scientific knowledge and technological power to formulate and execute the rehabilitation plan. The first three items invoke scientific knowledge as an important source of rehabilitation planning and execution; they are the theoretical underpinning of the entire process.

It appears that the world of post-mining rehabilitation (at least in Australia) has an obvious biodiversity-conservation focus motivated by recovering the 'lost' biodiversity patterns. On the other hand, the ultimate goal of post-mine rehabilitation from the point of view of mining-companies is the creation of a functioning, self-sustaining ecosystem (Mucina and Dobrowolski 2015). These are two different goals, underpinned by different scientific theories and sets of important trade-offs. We suggest that knowledge of the plant community assembly, especially the new approaches focusing on plant functional traits rather than taxonomy, should find serious consideration in setting new, realistic restoration targets, especially in recovery/rehabilitation of species-rich 'recalcitrant' ecosystems.

Approaches to restoration on laterite and deep sands

Alcoa approach

Rehabilitation of Mediterranean-type eucalyptus jarrah woodlands (grading into dense forests) on extremely nutrient-poor soils is in the Australian context always cited a success story of rehabilitation (Banning *et al.* 2011; Grant and Koch 2007; Koch 2007). Perhaps this multi-layered and species diversity-focused approach is depicted best in a scheme presented in Figure 13.8. The approach takes into consideration a number of important ecological drivers and incorporates elements of landscaping as well as 'reconstructing of community structure', aimed at the major, politically important target – species diversity. How the reconstructed vegetation would behave in terms of ecosystem services remains to be seen since the pace of the vegetation-dynamic processes in slowly evolving systems is in discordance with the life span of a researcher or research-funding span. The rehabilitation of the jarrah woodlands and forests is surely an excellent example of cooperation between science and post-mining rehabilitation practice.

Iluka approach

Mining rehabilitation occurs in sequence with ore removal for mineral sands mining: topsoil removal and storage (or 'direct return' of topsoil where possible to other areas being rehabilitated); overburden removal and storage; removal and wet separation (physical separation by density) of the ore; return of sand/clay/mixed tailings to the pit; once tailings are drained, reshaped to an appropriate landform design; return of overburden then topsoil with its critical seed bank; broadcast of collected seed; surface stabilization against wind erosion with a cover crop, native mulch (now discontinued) or temporary chemical sticking agent; in-fill planting from nursery propagated material; and finally, monitoring of the establishing vegetation. Broad groupings of local floristic communities found on similar soils/landforms to those recon-

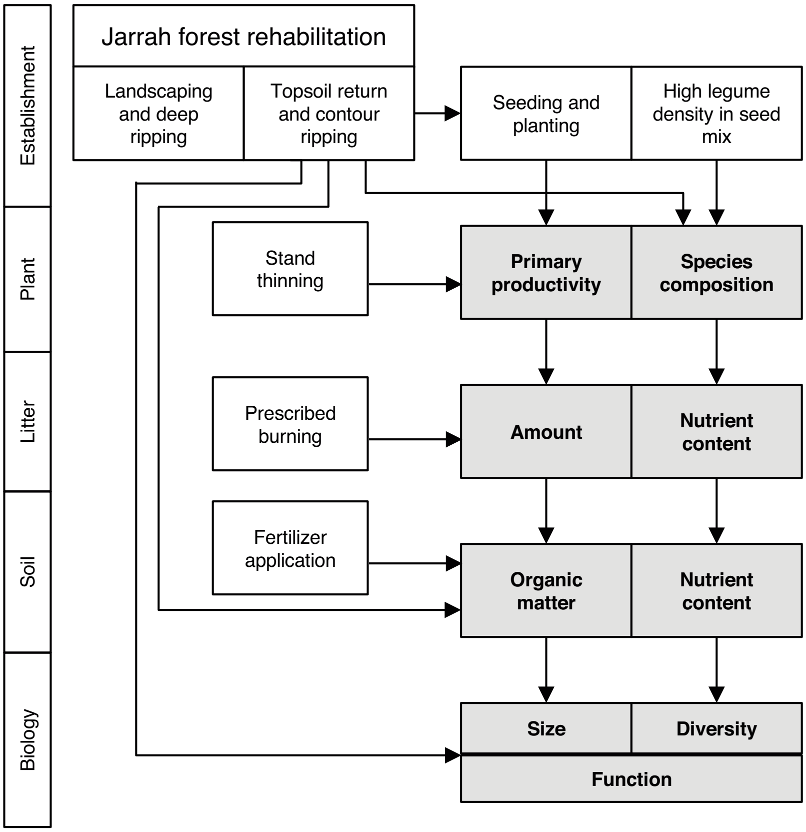

Figure 13.8 Schematic overview of the influence of management practices on ecosystem attributes in post-mining jarrah forest rehabilitation

Source: Banning *et al.* (2011)

structed in post-mining rehabilitation inform the choice of plant species from which seed is collected. This maximizes species diversity that can be selected in the knowledge that many species cannot be propagated from seed or in vegetative manner in nurseries.

New perspectives

Rehabilitation or restoration of species-rich systems showing a high level of functional complexity (rich spectrum of biotic interactions, specialized life histories of the constituent species, small-size populations, rarity etc.) is the ultimate frontier of the restoration science and practice.

Using the trait-focused approach to rehabilitation in predictive modelling of restoration and rehabilitation outcomes (Laughlin 2014) and its application in pot-mining rehabilitation should be broadened.

Acknowledgements

Laco Mucina thanks the Iluka Chair at the University of Western Australia for logistic support. This research was supported in part by an ARC Linkage grant LP150100339. Beatriz Duguy thanks A. Valdecantos and V. R. Vallejo for giving permission to reproduce some of their material, and in particular to V. R. Vallejo for his helpful comments and suggestions. Marcela A. Bustamante-Sánchez, Cecilia Smith-Ramírez and Juan J. Armesto thank project Fondef-Idea CA13I1027, project Native Forest Act 007, FONDECYT 11121452. Alberto Vilagrosa thanks projects SURVIVE (CGL-2011-30531-CO2-02), GRACCIE-NET (CTM2014-59111-REDC) and the PROMETEO program (DESESTRES 2014/038). CEAM is supported by the Generalitat Valenciana.

Credits

Laco Mucina co-authored (together with Mark Dobrowolski) the section on SW Australia and edited all sections and shaped the chapter. Beatriz Duguy Pedra and Alberto Vilagrosa co-authored the section on the Mediterranean Basin; Todd Keeler-Wolff wrote the section on Californian shrublands; Pat Holmes and Mirijam Gartner co-authored the part on the Cape (South Africa) shrublands, while Marcela A. Bustamante-Sánchez, Cecilia Smith-Ramírez and Juan J. Armesto co-authored the Chilean section of the chapter.

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