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The restoration of vegetation cover in the semi-arid Iberian southeast

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ABSTRACT

Semi-arid landscapes in the western Mediterranean have been used for millennia, resulting in large-scale transformations and widespread degradation. In some instances, these degraded environments have been unable to recover spontaneously, and ambitious restoration programs have been launched over the last decades to improve landscape conditions. Ecological restoration may speed up succession, promote more complex communities and increase their functionality. But uncertainty in the definition of restoration objectives, failure to identify most efficient practices and, particularly, socio-economic and cultural constrains may compromise future actions. Here, we review recent advances in the restoration of semi-arid vegetation cover in the Iberian southeast, discuss future challenges and suggest two key steps towards increasing the consistency and efficiency of restoration programs: emphasis on ecosystem services, and implementation of participative and adaptive management practices.

1. Introduction

The Iberian southeast is one of the major hotspots of biodiversity worldwide (Médail and Quézel, 1999). Unfortunately, it is also one of the most vulnerable areas to biodiversity loss and desertification (Fons-Esteve and Páramo, 2003). Both phenomena may be related, as human activity may be partly responsible for heterogeneous landscapes favoring diversification and land degradation (Blondel and Aronson, 1999). Humans have prospered in this area for millennia, and their impacts include deforestation, increases in wildfire recurrence, increases in conifer dominance, and erosion (Carrión et al., 2010). But humans are not the only cause of land degradation in the Iberian southeast. Natural factors such as scarce and irregularly distributed rainfall, erodible soils and steep terrain have all contributed to shaping current landscapes (Puigdefábregas, 1998). In the context of ecological restoration, this point is particularly challenging, as it recognizes the existence of 'naturally' degraded, dysfunctional or impoverished landscapes. Should these landscapes be 'restored'? Only when they compromise human welfare? Do these interventions qualify as 'ecological restoration'? These are not rhetoric questions, as desertification is the result of complex human–biophysical interactions, and causality is not always obvious.

In this review we describe how the aims and techniques used to restore the vegetation of degraded ecosystems in the Iberian southeast changed along the 20th century. We discuss current paradigms and present a new theoretical framework for the restoration of these areas. We then illustrate successful techniques to foster the restoration of plant cover based on biotic interactions and the use of technology. Finally, we emphasize the need to incorporate society aspirations and potential, to improve the success of restoration actions.

2. Restoration programs in the 20th century

Restoration objectives and the identification of target ecosystems depend on the ecological, socio-economical and cultural context. Thus, despite that the overall aim of restoration actions in the Iberian southeast has been the increase in plant (mostly forest)

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cover, particular restoration objectives, and the techniques to achieve them, have changed substantially throughout the 20th century. In the early days of the past century, a few large restoration projects were launched to reverse the unfortunate effects of the 19th century land use policies (Bautista et al., 2010). Those projects involved huge investments, lasted for several decades, and created some of the oldest forest areas in the Iberian southeast. They were considered successful by the time they were completed, a perception that was probably favored by (i) the long time span of the projects, which allowed trial and error approaches, (ii) the identification of clear achievable objectives (e.g., establishing a tree cover, fixing sand dunes), and (iii) the social impact of employment in economically depressed areas. It is worth mentioning that vintage practitioners developed and tested a wide range of restoration techniques, merging traditional and empirical knowledge, and establishing the basis of what has been later defined as adaptive management (Codorníu, 1901).

The rationale behind restoration projects changed dramatically over the course of the 20th century (Vallejo, 1996). Projects became short-lived, scattered over the region, and used gradually increasing energy, and technological and economic inputs. The array of species planted narrowed towards a few ones, mostly *Pinus halepensis* Mill. For example, in the semi-arid area of the province of Alicante, the Forest Administration carried out more than 200 plantation projects between 1940 and 1995, totaling 13,500 ha (2.4% of the total area), and used *P. halepensis* in 99% of them. These plantations now contribute to the vast area covered by this species in the Iberian southeast, as occurs in other Mediterranean areas (Barbéro et al., 1998). But failures to improve ecosystem function (Goberna et al., 2007), negative impact of pine plantations on other species (Bellot et al., 2004; Navarro-Cano et al., 2009), high mortality rates and slow growth (Maestre and Cortina, 2004b), although not widespread, gained social visibility, fostered criticism and favored a new shift in paradigm (Maestre and Cortina, 2004b; Padilla et al., 2009a,b).

By the end of the 20th century, it was clear that Clementsian successional trajectories, which were at the heart of restoration programs based on the extensive use of *P. halepensis* (e.g., Ruiz de la Torre, 1973), did not always work, and further actions were needed to establish the diverse functional and resilient landscapes identified as restoration target. There was an increasing trend towards using a wide array of species, protecting extant vegetation cover, making use of positive interactions, and preserving ecosystem function by developing new machinery, and reducing the extent of the interventions (Cortina et al., 2004; RECONDES Project team, 2010). Seed bank, nursery and field techniques gradually improved to accommodate a wide range of species, including shrubs, and to identify the most suitable species for particular habitats (Fig. 1; Cortina et al., 2004; Padilla and Pugnaire, 2009). Still, the establishment of plant cover was the objective of all efforts.

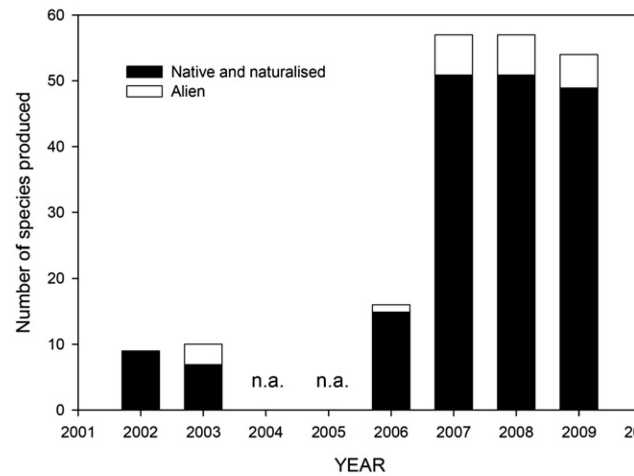

Fig. 1. Number of species produced by the three public forest nurseries in Alicante (Iberian southeast) in campaigns 2002 to 2009 (n.a.: unavailable data; F. Gil, Generalitat Valenciana, pers. comm.).

3. Limitations of current restoration programs

Some decades after the advent of the 'new' paradigm, characterized by the use of an increasing diversity of woody species and less aggressive plantation techniques, it is time to evaluate the outcomes of recent restoration actions. This task is challenging, as most of them are not monitored. Conversely, there has been a vast effort to study the effects of restoration techniques and improve their efficiency at an experimental scale. As in other areas of the Iberian Peninsula, this research has been strongly biased towards the establishment of woody species, and the response of individuals rather than populations and upper organization levels (Cortina et al., 2008). In addition, most studies cover less than 10 years, a time span that may be insufficient to fully evaluate the outcome of management practices in a semi-arid area.

Over the 20th century, restoration recognized the need to foster succession by promoting the establishment of late-successional species. The rationale behind the reintroduction of late-successional species was that they are important components of 'potential vegetation', a pythosociological term related to Clementsian climax (Rivas-Martínez, 1997). However, the use of 'potential vegetation' to identify restoration targets may be flawed, as 'potential vegetation' has been traditionally described from a small set of relatively undisturbed sites. Firstly, these sites may not represent the full range of conditions characteristic of degraded sites or the wide range of meta-stable states defined in state-and-transition models (Cortina et al., 2006a). Secondly, degradation may preclude the return to a pristine condition. Thirdly, 'potential vegetation' may be indeed the result of long-term use by humans, and thus it may not represent a pre-disturbed state (Badal et al., 1994). Finally, the environment is changing, and current species assemblages may not endure future conditions (Hobbs et al., 2009). This point has important implications for restoration in the Iberian southeast, as it is one of the driest areas in Europe and may be particularly susceptible to climatic changes (Giorgi and Lionello, 2008). For example, biogeographical envelop models suggest that species in this area may not withstand changes in climatic conditions, and there may not be other species in the European pool to replace them (Bakkenes et al., 2010). Unfortunately, predictions on future climatic conditions have been barely integrated into current restoration programs.

Despite its limitations, the 'potential vegetation' approach may be useful in identifying key species whose reintroduction should be prioritized. For example, the presence of resprouting shrubs in *Stipa tenacissima* steppes has been associated with high soil fertility, high animal and plant diversity, and the regulation of soil microbial communities (López and Moro, 1997; Maestre and Cortina, 2004a; Maestre et al., 2009). In addition, most of these species increase ecosystem resilience by resprouting after disturbance, and create new niches favoring the establishment of other species (Pausas et al., 2006). However, knowledge on the natural dynamics of key woody species and their interaction with other community components is rather weak. For example, current planting density (e.g., 400-700 seedlings ha^{-1}) may be substantially higher than the maximum density attainable by these species in natural landscapes (e.g., a maximum of 204 patches of resprouting shrubs above 1 m diameter per hectare in *S. tenacissima* steppes in Alicante; B. Amat, unpubl. data). Thus, high mortality may not reflect the failure to apply proper planting techniques but rather our inability to identify suitable planting microsites (Maestre et al., 2003b). This distinction is relevant, as in addition to planning costly irrigation programs, and improving seedling quality and soil preparation techniques, we should enhance our ability to find indicators of potentially successful planting areas and planting spots. It is worth noting that experiments aimed to evaluate the outcome of restoration practices tested in several replicated areas often show much higher variability associated with landscape heterogeneity than to the experimental treatment itself (Fig. 2). Furthermore, species incorporated into current restoration programs show an extremely wide range of spontaneous recruitment success, suggesting that some species may not need external assistance to establish, whereas others may need unaffordable inputs to get established (Rey and Alcántara, 2000; Padilla et al., 2009a,b). In both cases current planting programs would be misusing resources and efforts.

4. Beyond the use of 'potential' vegetation as a restoration target

Considering the difficulties in identifying late-successional species and the limitations of 'potential vegetation' to represent reference ecosystems, which should be the target of restoration programs? We suggest that restoration targets should encompass the wide range of communities suitable for a particular site, and should be defined in terms of ecosystem functioning and ecosystem capacity to provide goods and services, in addition to community composition (Fig. 3). Accordingly, the role of restoration practitioners should be (i) to identify potential states in state-and-transition models, and provide information on the composition, function, goods and services provided by each state, (ii) to rate the probability of transitions between current degraded state and alternative states, and particularly, to identify the most likely trajectory towards a desired state, and finally (iii) to select acceptable tools and protocols to promote the desired change. It is important to emphasize a precautionary principle that should drive the whole process: any measure taken should maintain or increase ecosystem resistance and resilience to further disturbances, and particularly, it should prevent crossing current and future irreversible degradation thresholds.

How can we apply this framework to the restoration of degraded ecosystems in the Iberian southeast? First, it must be emphasized that targets for the restoration of degraded areas are diverse, and they should be defined on the basis of current status and social needs. We may foresee a spatially heterogeneous land, with scattered trees (mostly *P. halepensis* and *Tetraclinis articulata*) and resprouting shrubs, forming patches in a matrix dominated by herbaceous species (e.g., *S. tenacissima* and *Brachypodium retusum*), and open areas. Large areas in the Iberian southeast are reasonably well conserved in terms of their composition, function and services, and may actually be recovering from past disturbances (Alados et al., 2004; Peña et al., 2007). In these cases, no intervention may be needed, unless the current state is not the desired state for the local population.

Restoration of degraded lands should aim at reversing the effects of degradation by improving soil conditions, increasing plant cover, and introducing key-stone woody species. In the Iberian southeast, restoration should follow a two-step approach according to their functional status and structural attributes (Maestre and Cortina, 2004a). In lands showing symptoms of impaired functionality, restoration actions should focus on repairing soil stability, infiltration and nutrient cycling. This can be achieved by using low-cost techniques such as the creation of resource sinks (Tongway and Ludwig, 1996; Soliveres-Codina et al., 2008)

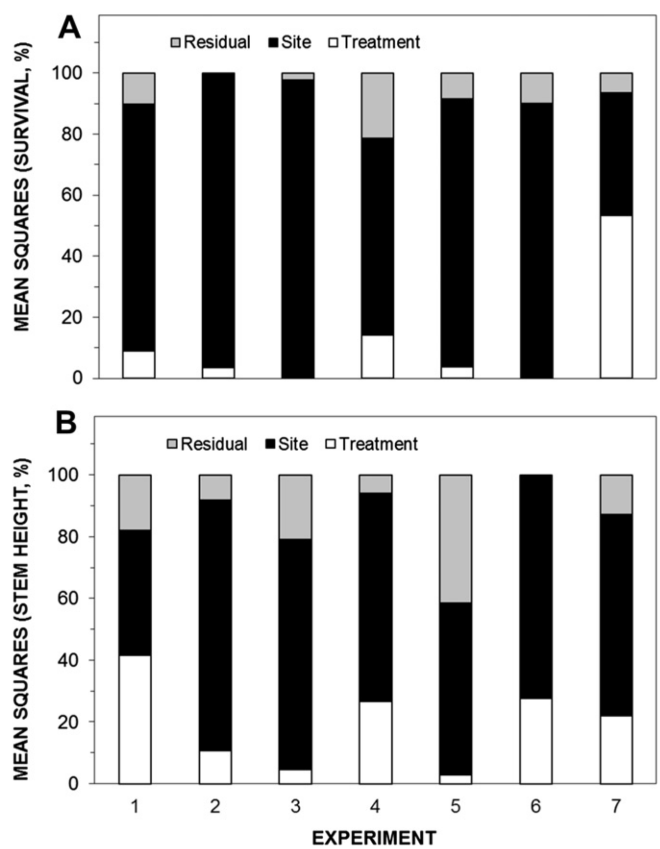

Fig. 2. Variance partitioning in 7 experiments measuring survival (A) and stem height (B) of *Pistacia lentiscus*, *Quercus coccifera* and *Juniperus oxycedrus* seedlings one summer after planting in the Iberian southeast. Experiments were performed in different years and involved different treatments, including facilitation (Experiment 1), site preparation (Experiment 2), drought preconditioning (Experiments 3–6) and acorn pre-germination (Experiment 7). Data from B. Amat (University of Alicante, unpublished; 1), D. Fuentes (Fundación CEAM, unpublished; 2), Fonseca (1999; 3–5), Rubio et al. (2001; 6) and Vilagrosa et al. (1997; 7).

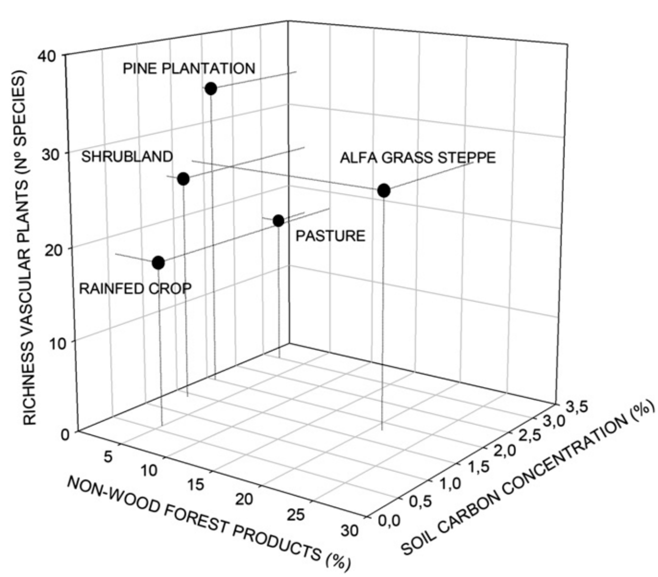

Fig. 3. Landscape units in a representative landscape of the Iberian southeast classified in a 3-D space defined by soil fertility (soil organic carbon content), the number of vascular plants and the frequency of species providing non-wood forest products as fiber and fruits (data compiled from different sources by M. Derak, Haut Commissariat aux Eaux et Forêts et à la Lutte Contre la Desertification, Tetouan, Morocco).

and plantation of highly stress-tolerant species (Padilla et al., 2009a,b), and may be more efficiently deployed by considering the spatial pattern of resource sinks and sources and the connectivity between them (Bautista et al., 2007; RECONDES Project team, 2010). It is important to note that strategic distribution of vegetation cover, and not the increase in vegetation cover to its maximum achievable, may maximize some functions (e.g., soil protection), while minimizing the amount of transpired water (Bautista et al., 2007).

In areas with higher functional status and higher potentiality (deeper soils, northern aspect), restoration actions should aim at introducing or reinforcing the population of large resprouting shrubs and trees as a way to improve ecosystem functions, to increase ecosystem resilience, and to foster colonization of other plant and animal species (Maestre et al., 2009; Padilla et al., 2009a,b). In degraded pine plantations, shrubs may be preferentially located in open areas, to avoid negative interactions with pines and accompanying herbaceous vegetation (Maestre et al., 2004). In addition, herbaceous populations may be reinforced to reach maximum plant cover according to site potential, and to minimize interpatch distance and the risk of resource leakage (Maestre and Cortina, 2004a).

By contrast to degradation thresholds, the concepts of restorability and restoration thresholds have received much less attention. Restoration thresholds may occur when restoration success and efforts are not linearly related, for example, when a given density of introduced individuals is needed to ensure persistence (Montalvo et al., 1997). Restoration thresholds are particularly relevant, as they may identify actions with particularly low cost-benefit ratios. For example, in *S. tenacissima* steppes, the introduction of large resprouting shrubs may have a synergistic effect on other components of the ecosystem (Maestre et al., 2009). Achieving a certain density of species such as *Quercus coccifera*, *Pistacia lentiscus* and *Rhamnus lycioides* may foster facilitative interactions and community assemblage at relatively low cost. The information available on the efficiency of different practices (i.e., on the advance towards a given target per amount of restoration effort) is, however, scarce.

5. Using biotic interactions to foster the restoration of plant cover

In semi-arid ecosystems of the Iberian southeast, vegetation is often arranged as a two-phase mosaic of vegetated patches inserted on a bare ground matrix (Puigdefábregas et al., 1999). The spatial distribution of vegetation cover favors the formation of 'resource islands' beneath the canopy of dominant plants such as *Retama sphaerocarpa* and *S. tenacissima*, increasing ecosystem productivity and diversity, and promoting plant establishment (Maestre et al., 2001; Goberna et al., 2007; Padilla and Pugnaire, 2009).

There are numerous techniques available to overcome limitations to plant establishment during restoration in semi-arid environments. Among them, the use of positive interactions between neighboring plants has received substantial attention in recent years (see Padilla and Pugnaire, 2006 for a recent review). Some of the pioneer work on this topic was carried out in *S. tenacissima* steppes and *P. halepensis* plantations from southeast Spain (Maestre et al., 2001, 2002, 2003a, 2004). In those experiments, seedlings of different woody species (*P. lentiscus*, *Q. coccifera*, *R. lycioides* and *Medicago arborea*) were introduced under the canopy of *S. tenacissima* and *P. halepensis* and in areas devoid of vascular plants. The results obtained were mainly dependent on the climatic conditions of the first year after plantation, and the benefactor/beneficiary species considered. *S. tenacissima* facilitated the establishment of the introduced seedlings in most cases, but this effect was not universal, and negative interactions between both *S. tenacissima* and *P. halepensis*, and the introduced seedlings were observed under high abiotic stress. A facilitative effect of *S. tenacissima* tussocks on germination has been observed in *R. lycioides* and *S. tenacissima* (Barberá et al., 2006). More recently, Padilla and Pugnaire (2009) compared seedling survival of *Olea europaea*, *P. lentiscus*, and *Ziziphus lotus* under the canopy of *R. sphaerocarpa* and in gaps covered with piled branches that mimicked the shrub canopy. As found in previous studies, survival of seedlings planted under *R. sphaerocarpa* differed depending on species identity and on the supply of additional water through irrigation.

Uncertainty in the direction and magnitude of the interactions is not the only factor challenging the use of facilitation at a management scale. Most experiments on the use of plant–plant interactions in restoration mimic natural conditions by digging small planting holes to avoid soil disturbance. But deep soil preparation, a practice that is barely compatible with the use of facilitation, commonly has a strong positive impact on plant performance. A potential solution resides in the use of soil preparation tools that allow deep rooting while preserving the conditions of the favorable microsite (Castro et al., 2002). Conversely, identification of the mechanisms involved in the interactions (generally microclimatic amelioration and improved soil fertility; Azcón and Barea, 1997; Maestre et al., 2003a) may help to mimic the function of true benefactors. Overall, these results show that the use of facilitative interactions have potential to improve the restoration of degraded semi-arid ecosystems, but the effectiveness of this technique will be largely dependent of the level of abiotic stress experienced after planting, our knowledge of specific plant–plant interactions and our ability to develop planting techniques that maximize these interactions.

Among other relevant soil features, biological soil crusts (BSCs) deserve special attention when dealing with the restoration of semi-arid ecosystems. BSCs are a prominent feature of semi-arid ecosystems in the Iberian southeast with key functions in maintaining ecosystem structure and functioning in this area (Maestre et al., 2011). To overcome current limitations to their establishment in the Iberian southeast, and to speed up recovery, *in situ* inoculation of soils with BSC components, such as cyanobacteria, has been employed in degraded semi-arid ecosystems, including *S. tenacissima* steppes from southeast Spain (Maestre et al., 2006; Bowker, 2007). Further studies are needed to develop suitable application techniques at a management scale, and to test the effectiveness of BSCs as a complement to traditional restoration approaches (Maestre et al., 2011).

6. Using technological tools to foster the restoration of plant cover

In the Iberian southeast, plant colonization is strongly limited by biotic and abiotic factors (see section 3). Abiotic constrains are likely to increase as temperatures increase and rainfall becomes more scarce and concentrated (Giorgi and Lionello, 2008). Limitations imposed by recruitment bottlenecks can be diminished by employing suitable restoration techniques. Ecotechnological tools commonly used in semi-arid environments have mostly focused on (i) improving plant material capacity to endure stress, (ii) deterring seed and seedling predation, and (iii) improving microsite conditions and resource availability (Cortina et al., 2004). In many instances, these techniques mimic the ecological interactions described in the previous section, and may be a way to reverse degradation.

Seed germination and the early phases of seedling establishment strongly limit plant colonization (Rey and Alcántara, 2000). Thus, it is not surprising that seedling planting, rather than seeding, is the preferred technique in the Iberian southeast. Nursery techniques aiming to produce high quality seedlings have substantially changed over the last decades (Peñuelas and Ocaña, 2000; Cortina et al., 2006b). Nowadays, nursery protocols have been adapted to nursery and species requirements. Most nurseries grow seedlings for 6–12 months, use forest containers and artificial substrates based on peat, and expose seedlings to the open air. These techniques have certainly improved plantation success (Cortina et al., 2006b). Yet, the identification of morpho-functional traits favoring seedling establishment is an ongoing discussion. In Mediterranean areas, bigger seedlings commonly show higher survival and growth when planted in the field (Villar-Salvador et al., 2004). But this relationship may be different in semi-arid areas of the Iberian southeast, where several studies have failed to find positive relationships between seedling size and field survival (Seva et al., 2004; Trubat et al., 2011). The reasons for this may be related to the balance between root and foliar surface area (Trubat et al., 2006), and to the ability of seedlings to produce deep root systems before the onset of summer drought (Padilla and Pugnaire, 2007). These results emphasize the difficulty in establishing strict regulations on the quality of seedling stocks, and the need to focus on nursery techniques promoting rooting ability rather than aboveground morphological traits.

In the previous sections, we have emphasized the importance of favorable microsites created by biotic and abiotic agents to promote seedling establishment. When favorable microsites are scarce, environmental conditions and resource availability can be artificially improved. Field techniques to improve seedling establishment commonly prioritize the increase in rootable soil volume, nutrient availability, runoff collection and water conservation, while controlling competition with extant vegetation. Among them, site preparation is probably the most powerful technique to improve seedling water status.

When semi-arid ecosystems are highly degraded, the rate of resource capture is reduced, and vegetation structure, as well as ecosystem function, may be lost at intermediate or long-term (Ludwig et al., 2000). If this occurs, the spatial heterogeneity of soil resources may be a key element in the regeneration of vegetation after disturbance. In this direction, Maestre et al. (2003b) found that the spatial patterns of seedling survival in a plantation of *P. lentiscus* in a very degraded area of Alicante (southeast Spain) was strongly coupled to small-scale spatial patterns of bare soil cover, sand content, and soil compaction. These results illustrate how the identification of environmental factors associated with mortality clumps can guide restoration efforts; areas of potential high mortality could be defined beforehand by performing an analysis of the spatial distribution of those variables more related with seedling survival before planting. Alternatively, restoration success could be improved by modifying those variables at the moment of planting.

Site preparation is commonly carried out by digging 40 × 40 × 40 cm planting holes or subsoiling at 50–60 cm depth using mechanical means. These techniques aim at increasing available soil volume, runoff capture, moisture conservation, and soil water holding capacity. Facilitating deep rooting may be more important than increasing the volume of altered soil (Padilla and Pugnaire, 2007). Several studies have shown that deep soil preparation increases seedling survival and growth (Querejeta et al., 2001; Barberá et al., 2005). It is worth noting that site preparation techniques have gradually gained in efficiency while being increasingly gentle with extant vegetation and remaining ecosystem functions (Appendix 1). Many of the drawbacks of late 20th century plantations have been solved in this way.

Soil preparation can be completed by creating manually-built structures that may increase water availability in the planting hole. In the Iberian southeast, as in other arid areas, simple techniques based on traditional knowledge, such as micro-catchments, represent efficient approaches to increase the survival and growth of planted seedlings (Fuentes et al., 2004). But the effectiveness of these techniques is restricted to rainfall events generating surface runoff, while light rainfall events (1–10 mm) are most frequent in the Iberian southeast, particularly in summer (Lázaro et al., 2001). These inputs are largely lost by interception and evaporation (Domingo et al., 1999). Various techniques have been developed to avoid these losses by increasing runoff concentration and enhancing water infiltration (Li et al., 2000).

Degraded soils are frequently infertile, and planted seedlings may respond to additional nutrient inputs (Cortina and Maestre, 2005; Valdecantos et al., 2006). Organic amendments represent a suitable source of nutrients and organic matter that can be used in forest plantations. Organic amendments may increase soil water storage capacity and ameliorate seedling nutrient status (Querejeta et al., 2001; Fuentes et al., 2010). A single application of composted sewage sludge at planting at relatively low dose (e.g., 15–30 Mg dry weight ha^{-1}) may sustain seedling response for several years (Fuentes et al., 2010). Higher doses (e.g., >100 Mg ha^{-1}) may also trigger seedling response (Querejeta et al., 2001; Barberá et al., 2005), but they should be restricted to deep soils and combined with intense soil preparation to avoid negative effects of high application rates such as increased osmotic stress (Fuentes et al., 2007). Furthermore, the application of organic amendments should be made with care to minimize the negative effects of increased competition with extant vegetation (Querejeta et al., 2008; Fuentes et al., 2010), uncertainties on the fate of allochtonous microflora and xenochemicals, and their impact on soil biota (Gómez-Rico et al., 2008), and increased costs (Valdecantos et al., 2004). As a result of these constrains, organic amendments are barely used at a management scale.

Unlike other arid and semi-arid areas (Bainbridge, 2002), irrigation has been seldom used in restoration projects in the Iberian southeast. But this practice may change in the future as restoration projects are increasingly including emergency watering in their plans. A large number of irrigation systems are available to deliver water to planted seedlings in an efficient way, but scientific literature on this topic is scarce. Small water inputs applied by drip irrigation may be sufficient to dramatically decrease seedling mortality (Sánchez et al., 2004; Padilla et al., 2009a,b).

Water sources other than rainfall may play a relevant role in the Iberian southeast (Ramírez et al., 2007). A network of occult precipitation gauges, together with modeling are being used to map areas where fog harvesting may provide an additional source of water for plantations (Estrela et al., 2008). Collectors have proven particularly successful in capturing fog water, with peak rates above 10 L m^{-2} day^{-1} (Estrela et al., 2009). Designing collectors so they can feed individual plants (del Campo et al., 2006), represents a challenge, as they would reduce the costs of deployment and maintenance of irrigation systems.

7. Socio-economic constrains and opportunities

Ecological restoration, in the way it has been previously defined, is a human construct. Society decides, by act or omission, what is expected from a degraded site, and how much effort should be committed to bring the system towards a desired state. Desired state is usually defined in terms of a fuzzy combination of composition, functionality and provision of services. Thus, it is important to note that the desired state may not always be the optimum from a biophysical point of view; and it is often not. Prioritizing anthropically-biased solutions, which may even put ecosystem sustainability at risk, would be as reckless as imposing biophysical states that do not take into account society needs and aspirations. Unfortunately, merging biophysical and sociological (including cultural and economical) goals is not an easy task. Research has strongly focused on biophysical aspects, and this bias has major consequences on our ability to improve restoration success.

Most studies on the design and evaluation of novel restoration techniques have been implemented at an experimental scale, which is commonly below the scale that is relevant for management. Results from experiments can hardly be transferred directly to managers, as constrains and priorities may differ in a way that may compromise restoration success. For example, the geographical scale of experiments and restoration projects often differ, and a solution that may be suitable for a 30×30 m plot may reveal unpractical when applied at a larger scale. Watering or the application of organic amendments is frequently unfeasible on rugged and remote terrains with no access to machinery, despite that this practice may rend good results when applied at a small experimental scale. Costs of a given technique may also become unaffordable when increasing the surface area affected, and considering the contrasted budgets of restoration experiments and restoration actions. Experiments commonly last for a few years and focus on specific processes and ecosystem components, which is insufficient to fully capture the applicability and impact of a new restoration technique. It is particularly important that research on ecological restoration will gradually integrate a landscape perspective, e.g., by identifying priority areas that may act as corridors, and reducing the connectivity of open patches (Kuijken and de Blust, 2003). Finally, experiments are commonly reductionistic, whereas restoration projects are holistic by nature (White et al., 2002). Our previous discussion on the use of positive plant–plant interactions in restoration practice is a good example of the difficulties to upscale research results to a management level.

The gap between experiments and projects hampers the transfer of research results to managers and the use of realistic scenarios in scientific studies. Clearly, there is a need of intermediate level studies, in the form of pilot projects, which may help to validate successful restoration practices at a management scale. Pilot projects could be incorporated in restoration programs as part of an adaptive management strategy (Pattel et al., 2007). With this aim, a pilot project to test the best available restoration practices at a management scale was established in Albatera (Alicante) in 2004 (Vilagrosa et al., 2008). This project made a thorough analysis of landscape state prior to the intervention, identified different landscape units and assigned selected species and technological tools to them. It used high quality seedlings and tested soil preparation techniques, such as 60 cm deep planting holes dug with backhoe spider, micro-catchments, organic amendments and mulches. Nowadays, the results are clearly visible, and the area has been extensively used for teaching and outreach activities.

Unfortunately, rigorous evaluation and monitoring programs are scarce. A joint effort from researchers, practitioners and policy makers is needed to incorporate them into restoration programs, and implement adaptive management strategies to (i) identify gaps in our knowledge on successional trajectories, restoration targets, and tools to achieve them, (ii) establish well designed experiments taking advantage of the enormous potential of restoration programs, (iii) monitor the outcomes of restoration programs in the long-term, and (iv) use this information to design alternative actions to redirect ecosystem change towards desired states. The implementation of ambitious monitoring programs and sound analyses should be priorities for the restoration of degraded ecosystems in the Iberian southeast.

The identification of desired ecosystem states should be the result of a cooperative effort from researchers (or whoever has the information), practitioners and other stakeholders. This exercise is not simple, as the decision must be based on a number of criteria that do not covariate and whose priority is not always clear. For example, under dry sub-humid conditions in southeast Spain, shrublands dominated by obligate seeders such as *Rosmarinus officinalis* show higher levels of diversity of vascular plants than *Pinus pinaster* forests, but biomass accumulation is lower in the former, generating a compromise between biodiversity and C sequestration (Cortina et al., 2006a). Trade-off interactions in ecosystem services are common, and may also occur in *S. tenacissima* steppes and *P. halepensis* plantations in semi-arid areas (Cortina et al., 2006a; Boix-Fayos et al., 2007). Multiple criteria decision models provide excellent tools to deal with this type of conflicts (Díaz-Balteiro and Romero, 2008). Yet, they have been barely used to evaluate alternative targets of restoration programs in semi-arid areas.

Restoration programs in the Iberian southeast have been traditionally carried out by the Forest Administration. Social involvement is commonly restricted to the provision of labor. There are many reasons for this, including the perception that ecological restoration is not a priority in rural areas, gradually absorbed by the tertiary sector (Montiel-Molina, 1990), and the absence of cost-benefit analyses that could provide evidence of the benefits supplied by restoration actions. Thus, it is no surprising that population is commonly unaware of restoration programs and their benefits. This represents a clear limitation for their success. We can envisage several solutions to these problems. On the one hand, the role of local communities, environmental NGO's and other private stakeholders has increased in the last decade, and this trend will probably continue if the Iberian southeast follows the trajectory defined by more developed regions. These initiatives are usually smaller than those lead by the Administration, and the socio-economic and technical context is substantially different. On the other hand, resources for ecological restoration have been gradually decreasing. It may be worth mentioning that the cost of a regular plantation in the Iberian southeast is ca. 3000 € ha^{-1}, depending on the species and planting techniques used, or 42% of the costs of a student in a public University (7172 €; Fundación EROSKI, 2009). As development and Common Agriculture Policy funds have shrunk in this area, it is not obvious where funding for future restoration actions will come from. But restoration can be a source of human welfare and, as such, it should be included in development programs for rural areas. By integrating restoration with complementary actions on the environment, economy, health and education of these areas, restoration could be sustained and could contribute to their development. The EU Rural Development Regulation (Council Regulation [EC] 1685/2005), the Spanish Law for Sustainable Development of Rural Areas (Law 45/2007), the European Regional Development Fund and Cohesion Fund, and the forthcoming EU strategy on Green Infrastructure could contribute in this respect (Hernández and Romero, 2009). Finally, there is a need to quantify the benefits of restoration by estimating their economic value (Hurd, 2009) or by implementing collective choice approaches, whereby the value of restoration actions is defined by a thorough understanding of ecosystem services and stakeholder negotiation (Shabman, 1995).

8. Concluding remarks

Over the last century, a huge amount of effort has been directed towards the restoration of degraded landscapes in the Iberian southeast. But the goals of restoration programs have changed as scientific knowledge built up, and society perspectives and aspirations evolved. Nowadays, restoration programs aim at reintroducing key species and foster successional trajectories towards an ideal state, often identified as 'potential' vegetation. We suggest that restoration programs should integrate state-and-transition models, and identify desired trajectories and states on the basis of community composition, ecosystem functioning and the provision of goods and services.

We have made substantial advances in the design and use of ecotechnological tools to foster the restoration of plant cover. However, too often the outcome of restoration programs is uncertain. This may partly result from scarce knowledge on species requirements, and the inability to identify suitable sites for planting and selecting the ecotechnological tools needed to overcome limitations for plant establishment. The identification of restoration threshold and the prescription of efficient ecotechnological tools will certainly improve the success and predictability of restoration programs. Pilot and demonstration projects, integrated into adaptive management programs, could provide further help in this direction.

Finally, society should participate in all phases of restoration programs, as they provide excellent opportunities to improve human welfare. This objective may be achieved by encouraging true participative management, and integrating restoration actions into land planning and development programs.

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Appendix. Supplementary data

Supplementary data associated with this article can be found, in the online version, at doi:10.1016/j.jaridenv.2011.08.003.

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