Innovations in Semiarid Land Restoration. The case of Stipa tenacissima L. Steppes

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Introduction

Stipa tenacissima L. (Alpha grass) steppes cover 32,000 km^2 in the western Mediterranean basin. These are the remains of an estimated 86,500 km^2 area covered by this species some decades ago (Le Houérou 1995). Reduction in *S. tenacissima* cover results from a combination of factors, including adverse climatic periods and changes in land use, especially overgrazing. *Stipa tenacissima* steppes are mostly distributed in a thin latitudinal fringe in North Africa, from Libya to Morocco, and in the southeastern portion of the Iberian peninsula, where they cover ca. 6,000 km^2.

Stipa tenacissima is of Asiatic origin, and it probably arrived in the SE Mediterranean during the Messinian crisis, 6.5 to 5.0 million years ago (Blanco et al. 1997), when large parts of the current Mediterranean basin dried out. Later, expansion was favoured by humans, as they removed accompanying woody vegetation (Barber et al. 1997, Buxó 1997). There is evidence of deforestation occurring in the area as early as the Copper Age (prior to 4,000 years BP), and artifacts made from *S. tenacissima* leaves, such as baskets, strings and shoes, dating back 3,000 years BP have been found in archaeological sites (Díaz-Ordoñez 2006). *Stipa tenacissima* steppes occupied vast extensions in southeastern Spain during Roman times (1st century): a dry area estimated at 50 x 150 km close to Carthago Nova (currently Cartagena) was named 'campus spartarius' (literarily meaning 'esparto grass field') by Pliny the Elder (Blanco *et al.* 1997). The importance of *S. tenacissima* for weaving and high quality paper paste increased up to the early 20th century, and there are records of local shortage of *S. tenacissima* fiber as early as 1879 (Hernández 1997). Several government agencies were created in the mid 20th century to foster this crop, such as the National Esparto Grass Service (Servicio Nacional del Esparto; 1948), launched by the Ministry of Industry and Commerce and the Ministry of Agriculture. However, the use of plastic fibers and rural exodus forced a sharp decline in *S. tenacissima* cropping, that almost disappeared by the end of the century. *Stipa tenacissima* is still used in North Africa, where it provides pastures and fiber for paper mills (e.g., as much as 250,000 Mg of raw cellulose and high quality paper paste in Kasserine factory in Tunisia, and Baba Ali and Mostaganen in Algeria; El

Hamrouni 1989), and its cropping and marketing may be fostered in SE Europe due to renewed interest in natural resources and traditional handcrafts which employ them.

In their present state, *S. tenacissima* steppes in SE Spain are the result of a long-term use of formerly wooded steppes that included cutting, burning and overstocking (Le Houérou 1995), followed by abandonment. Open forests of *Pinus halepensis*, *Tetraclinis articulata*, and *Pistacia lentiscus* with *S. tenacissima* have apparently been degraded to the state of tall shrublands dominated by *Rosmarinus officinalis*, *Pistacia lentiscus*, and *Phyllyrea angustifolia*, in the past. Further pressure on these ecosystems resulted in *S. tenacissima* steppes, eventually supporting fragments of the pre-existing, less disturbed vegetation. Woody vegetation was intentionally eliminated from *S. tenacissima* exploitations to reduce competition with the grasses. Improved *S. tenacissima* performance in the absence of *Pinus halepensis* has indeed been demonstrated (Gasque and Garcia-Fayos 2004). In some areas, overexplotation of *S. tenacissima* steppes may have favoured dwarf-shrubs at the expense of tussock grasses such as *S. tenacissima*, and a resulting decrease in plant cover (Maestre and Cortina 2004a).

Ibero-North African steppes are rich in endemic species. For example, close to 20% of the vascular plants in Spanish and North African steppes are endemic. But suppression of woody vegetation has probably had a strong impact on the abundance of vascular plants in *S. tenacissima* steppes (Maestre and Cortina 2004a). It is worth noting that biological crusts formed by mosses, lichens and cyanobacteria, are common in these steppes. These may also show high diversity (e.g. >15 taxa of cyanobacteria in 22 cm^2; Maestre et al. 2006a).

Stipa tenacissima steppes constitute an excellent model ecosystem to expand our knowledge of ecosystem dynamics in semi-arid lands because of their broad geographical distribution and their strong and long-term links with human activities. In addition, the wide variety of conditions characterising *S. tenacissima* steppes make this ecosystem particularly suitable to test the theoretical background of restoration ecology, and explore new approaches for the restoration of semi-arid areas. In this chapter we describe the main features of *S. tenacissima* steppes, particularly those from SE Spain, and suggest a framework for the restoration of semi-arid ecosystems. We use this framework to discuss the restoration of *S. tenacissima* steppes, and to describe ecotechnological tools based on existing knowledge of their dynamics and functioning.

Water as a main driver of ecophysiological responses

Stipa tenacissima commonly grows on shallow soils between the 200 and 400 mm annual rainfall isohyets (Sánchez 1995, Barber *et al.* 1997), but it can be found above and below these limits (Boudjada 2003, Le Houérou 1995). It presents several morpho-physiological adaptations to resist water stress (Table 1). It has been suggested that *S. tenacissima* behaves as an "opportunistic" species, as it can respond rapidly to short water pulses, such as late summer rainfall events (Pugnaire et al. 1996). Plasticity to nutrient availability may also be high (Pugnaire and Haase 1996).

TABLE 1. Physiological traits of *Stipa tenacissima* leaves under low and high water stress conditions (from Pugnaire and Haase 1996, Pugnaire et al. 1996, and Balaguer et al. 2002). ψ: leaf water potential, RWC: leaf relative water content, Chl_{a+b}: a+b chlorophyll concentration, A: net photosynthesis, g_t: total water vapour conductance, Fv/Fm: photochemical efficiency of PSII, n.a.: not available.

Variable	Time of the day	*Minimum Water stress*	*Maximum Water stress*
ψ (- MPa)	dawn midday	1.0 - 0.54 2.3 - 1.21	< 8.5 - 5.0 n.a.
RWC (g g^{-1})	dawn midday	0.87 - 0.93 0.85 - 0.91	0.50 - 0.78 0.72 - 0.75
Chl_{a+b} (mg g^{-1})	seasonal average	0.842	0.298
Chl_{a+b} (mmol m^{-2})	seasonal average	542	251
A (mmol CO_2 $m^{-2}s^{-1}$)	dawn midday	8.83 - 15.7 4.75 - 14.6	0.0 - 7.09 -1.6 - 3.89
g_t (mol H_2O m^{-2} s^{-1})	dawn midday	0.13 - 0.22 0.08 - 0.22	0.07 - 0.08 0.04 - 0.05
Fv:Fm	dawn midday	0.69 - 0.80 0.35 - 0.69	0.20 - 0.64 0.05 - 0.57

Stipa tenacissima leaves are 30-100 cm long, and they are produced from a tiller at a rate of 2.4-2.7 leaves $tiller^{-1}$ $year^{-1}$ (Sánchez 1995). Elongation rate ranges between 4 and 5 mm day^{-1} (Haase et al. 1999). *Stipa tenacissima* leaves are spatially arranged in a way that self-shading prevents photoinhibition, reduces as much as 40% of carbon gain, and increases water use efficiency at high water availability levels (Valladares and Pugnaire 1999, Ramírez 2006). Leaf senescence occurs mainly in summer (Table 1). Balaguer et al. (2002) emphasized *S. tenacissima* capacity to reverse leaf senescence following mild water stress. Reverse senescence, however, may be impaired in young individuals, thus compromising recruitment through sexual reproduction (Ramírez 2006). Productivity and biomass accumulation of *S. tenacissima* steppes are highly variable, and dependent on water availability and grazing pressure (Table 2).

TABLE 2. Biomass accumulation of *Stipa tenacissma* steppes in semiarid areas of SE Spain and North Africa.

Location	Biomass accumulation (Mg ha^{-1})	Reference
Rogassa, Algeria (grazed)	0.25-1.5	Aïdoud (1988)[1]
Rambla Honda, Almería	12.9	Sánchez (1995)
NW Algeria (grazed)	0.17	Debouzie et al. (1996)
Baza basin, Granada	7.8	Gauquelin *et al.* (1996)
Rambla Honda, Almería	3.6-4.8	Puigdefábregas et al. (1997)
Alluvial fan sector –Rambla Honda, Almería	1-1.5	Puigdefábregas *et al.* (1999)
Open hillslope –Rambla Honda, Almería	3.5-4.5	Puigdefábregas *et al.* (1999)

[1] In Le Houérou (1995).

The average life of a leaf is 2.5 years (Sánchez 1995). Dead leaves remain attached to the tussock for many years (e.g. 5.9 years; Sánchez 1995), forming a dense necromass layer that affects the growth and spatial arrangement of the whole tussock, and its capacity to form islands of fertility (Puigdefábregas and Sánchez 1996). *Stipa tenacissima* tussocks are cropped by pulling the leaves, and necromass accumulation is thus prevented in harvested stands.

Vegetative reproduction is considered to be *S. tenacissima*'s main space colonisation strategy (White 1983, Haase et al. 1995). This mechanism comes into play during the aging stage of the tussock, which is called the "degenerative senescent" phase, at around 60 years onwards (Servicio del Esparto 1950). In this phase, the tussock, which had gradually died from the centre, split into new tussocks in the periphery of the original individual (Puigdefábregas and Sánchez 1996). However, Gasque (1999), and Gasque and García-Fayos (2003) showed that *S. tenacissima* is capable of forming soil seed banks which depend on spike density. Seed germination and survival are related to vegetation density, water resources, and soil properties. A high degree of post-dispersal predation on *S. tenacissima*'s soil seed bank, in particular by seed-harvesting ants (1,500 seeds removed colony^{-1} day^{-1}), has been reported (Haase et al. 1995). Gasque (1999), however, considers that predation does not compromise recruitment in *S. tenacissima* grasslands because of the polyphagus nature of these seed-harvesting ants (*Messor* genus), which are capable of selecting and collecting the seeds of other plant species. *Stipa tenacissima* seeds are predated by other organisms including the bird *Bucanetes githagineus* (Trumpeter finch; G. López, pers. com.).

Stipa tenacissima roots constitute most of the biomass of the whole plant (61 %), but they do not grow deep into the soil (e.g., maximum rooting depth of 50 cm; Sánchez 1995). Shallow root systems enable *S. tenacissima* to respond rapidly to small changes in water availability (Domingo et al. 1991). Most of its rooting system is located beneath the tussocks, with little root volume colonising inter-tussock areas (Gauquelin et al. 1996, Puigdefábregas et al. 1999). This suggests that *S. tenacissima* has low capacity for capturing resources directly from open areas (Puigdefábregas and Sánchez 1996).

Ecological interactions in alfa grass steppes, from microscopic to landscape scales

Stipa tenacissima steppes are commonly structured in a spotted or banded spatial configuration (Puigdefábregas and Sánchez 1996, Webster and Maestre 2004, Maestre et al. 2005a), with vegetated patterns resembling features of the "tiger-bush" vegetation described for semiarid regions in Australia, the Sahel, Mexico and USA (Tongway et al. 2001). In some areas, this spatial configuration may be inherited from the spatial patterns favored by cropping techniques. But attributes of *S. tenacissima* patches, such as spatial pattern and cover, are crucial to maintain ecosystem structure and functioning (Fig. 1; Maestre and Cortina 2004a). In *S. tenacissima* steppes, the maintenance of vegetated patches is largely dependent on the redistribution of water, sediments and nutrients from the open areas to the discrete plant patches

(Puigdefábregas et al. 1999). Such redistribution may be influenced by topographical features (Puigdefábregas *et al.* 1999), by ecosystem structural attributes such as the number, width and spatial pattern of discrete plant patches (Imeson and Prinsen 2004), and by the soil surface conditions in the bare ground areas (Cerdá 1997, Maestre *et al.* 2002a). Thus, reductions in runoff fluxes reaching the plants promoted by changes in ecosystem structure or in soil surface conditions in the open ground may negatively affect the performance of *S. tenacissima* tussocks (Puigdefábregas *et al.* 1999; Maestre and Cortina 2006). Such a negative effect may ultimately modify ecosystem structure, impair its functionality, and promote degradation and desertification processes (Aguiar and Sala 1999). These "source-sink" dynamics, however, depend on complex interactions between climate, topography, vegetation and soil surface properties, and they are highly heterogeneous (see below).

Stipa tenacissima tussocks modify the availability of resources such as light, nutrients and water in semi-arid steppes at different spatial scales (Puigdefábregas et al. 1999, Maestre et al. 2001, 2003a, Ramírez 2006). Recent studies have thoroughly described the effect of these tussocks on their own microenvironment through the amelioration of the microclimate (Valladares and Pugnaire 1999), the improvement in the soil structure and depth (Bochet et al. 1999; Puigdefábregas et al. 1999), and the increase in soil moisture (Puigdefábregas and Sánchez 1996, Maestre et al. 2001), water infiltration (Cerdá 1997, Cammeraat and Imeson 1999), and carbon and nitrogen storage (Martínez-Sánchez et al. 1994, Sánchez 1995, Bochet et al. 1999) in relation to adjacent areas devoid of vascular plants. *Stipa tenacissima* creates "hotspots" of favorable soil conditions and microclimate, the so-called "resource islands" or "islands of fertility" a phenomenon commonly described in shrub species from arid and semi-arid areas worldwide (Whitford 2002). Interestingly, some studies have failed to observe changes in soil properties in the vicinity of *S. tenacissima* tussocks (M. Goberna and P. García-Fayos, pers. com.), suggesting that its capacity to generate resource islands may depend on site conditions.

Through the creation of resource islands, *S. tenacissima* modify the small-scale distribution and performance of a wide variety of taxa, such as biological soil crusts (BSC) organisms, soil fauna and vascular plants. In semiarid steppes of SE Spain, the distribution of *S. tenacissima* modifies the composition of BSC, with mosses dominating in the vicinity of the tussocks and cyanobacteria and lichens dominating the bare ground areas located between tussocks (Martínez-Sánchez et al. 1994, Maestre *et al.* 2002a, Maestre 2003). At larger spatial scales (e.g. 50 x 50 m plots), a positive association between the spatial pattern of *S. tenacissima* tussocks and that of BSC-forming organisms has been found (Maestre and Cortina 2002). Maestre et al. (2002a) found a negative relationship between the cover of cyanobacteria, which dominate bare-ground areas, and infiltration rate, suggesting that the effect of *S. tenacissima* on the composition of BSC-forming organisms could have relevant functional implications for the source-sink process described above, and thus for the own maintenance of *S. tenacissima* individuals. However, the relative importance of these organisms against other soil surface properties (microtopography and earthworm casts) and physical properties (texture and structure) as drivers of this process still needs to be addressed.

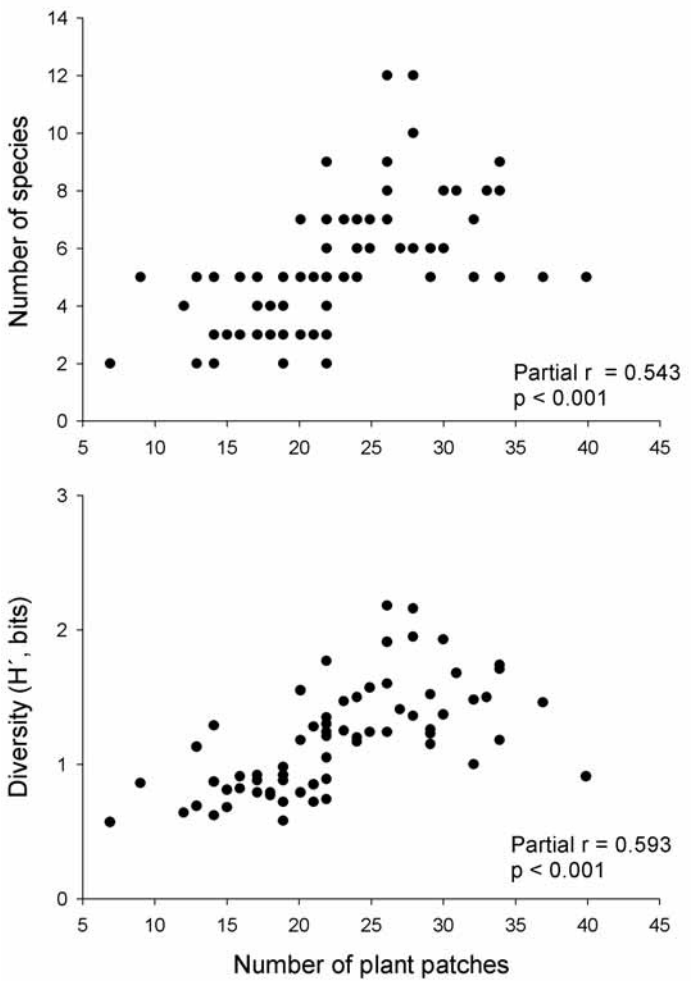


FIGURE 1. Relationships between the number of discrete plant patches and the richness and diversity of perennial vascular plants in semi-arid steppes of SE Spain. Results of partial correlation analyses, where the effect of plant cover is controlled, are shown in the lower right margin of each graph. Adapted from data summarized in Maestre and Cortina (2004a).

Regarding the relationships between *S. tenacissima* and vascular plants, the net outcome of their interaction –either facilitation or competition– depends on the scale considered, the identity of the species involved and prevailing environmental conditions. At small spatial scales, the vicinity of *S. tenacissima* tussocks holds more diversity and abundance of annual plants that the adjacent open ground areas (Sánchez 1995). Further, survival of seedlings and adults of woody species, such as *Pistacia lentiscus*, *Pinus halepensis* and *Quercus coccifera*, is higher in the vicinity of *S. tenacissima* tussocks than in open ground areas, indicating that *S. tenacissima* facilitates the establishment of these species (Maestre et al. 2001, 2003a; García-Fayos and Gasque 2003, Gasque and García-Fayos 2004). The amelioration of harsh climatic conditions through shading, as well as increase in soil fertility, has been identified as the main driver of facilitation (Maestre et al. 2003a). This effect, however, varies with the degree of abiotic stress

(Maestre and Cortina 2004b), illustrating the difficulties in fully understanding the net balance of plant-plant interactions.

Interactions between *Stipa tenacissima* and vascular plants have also been studied beyond the scale of discrete plant patches. Observational studies employing spatial analyses have reported positive and negative relationships between the spatial patterns of *S. tenacissima* and those of species such as *Anthyllis cytisoides* and *Globularia alypum*, the magnitude and even the direction of such relationships being dependent on the scale of the observation (Webster and Maestre, 2004; Maestre et al. 2005a). Gasque and García-Fayos (2004) compared the performance of seedlings and adult individuals of *S. tenacissima* in stands with and without *Pinus halepensis*. These authors found that *Pinus halepensis* had a negative effect on the reproductive output of *S. tenacissima* tussocks, as well as on the emergence, survival, and growth of seedlings of these species. They interpreted these negative effects as a consequence of rainfall interception by *Pinus halepensis*.

It is interesting to note that, at the stand scale, the cover of *S. tenacissima* has been negatively related to the diversity of vascular plants (Alados *et al.* 2006; Ramírez 2006). These results, which have been interpreted as the outcome of competition by *S. tenacissima* (Alados *et al.* 2006), do not agree with facilitative interactions reported at the tussock level (see above), and may rather result from historical removal of potential competitors in *S. tenacissima* crops. The mechanisms and consequences of the differential effects of *S. tenacissima* on other plant species at different spatial scales have not been fully addressed yet.

A framework for Stipa tenacissima steppes restoration

Ecosystem integrity can be defined in terms of ecosystem structure[1], function, and on the basis of the goods and services that they provide to human populations, including cultural values. Accordingly, ecosystem degradation reflects the loss of some or all of these components. Changes in ecosystem traits may not be harmonic: the loss of some species may not necessarily translate into proportional losses in ecosystem function, whereas some ecosystem functions may be impaired at relatively high species richness (Cortina et al. 2006).

Usually, degradation sequences do not follow a steady, gradual progress, but rather show phases of relative stability followed by phases of sudden change. Theses are known as degradation thresholds or transition boundaries, and may be first biotic (resulting from the loss of particular species), and then abiotic (e.g. resulting from intense deterioration of the physical environment; Milton 1994, Whisenant 1999). Thresholds are particularly relevant for restoration, as they represent ecosystem changes that may not be spontaneously reversible, or that may reverse at a rate that is slower than society demands (Bradshaw 2002).

Remnants of native, late-successional, sprouting shrubs like *Pistacia lentiscus* and *Quercus coccifera* play key functional and structural roles in semiarid *S. tenacissima* steppes. They

1. Here the term ecosystem structure includes both physical structure and biologial structure (i.e. species composition).

enhance ecosystem functioning (Maestre and Cortina 2004a), are a major determinant of plant diversity (Maestre 2004, Maestre and Cortina 2005), provide resilience against disturbances (Trabaud 1991), and supply shelter and food for wild and game animals (López and Moro 1997). The loss of these species represents a first degradation threshold in *S. tenacissima* steppes. The loss of *S. tenacissima* dominance, the loss of plant cover below ca. 30% (Thornes 1987; Thornes and Brandt 1994), and intense soil denudation may represent further steps in a degradation sequence in these ecosystems (Fig. 2).

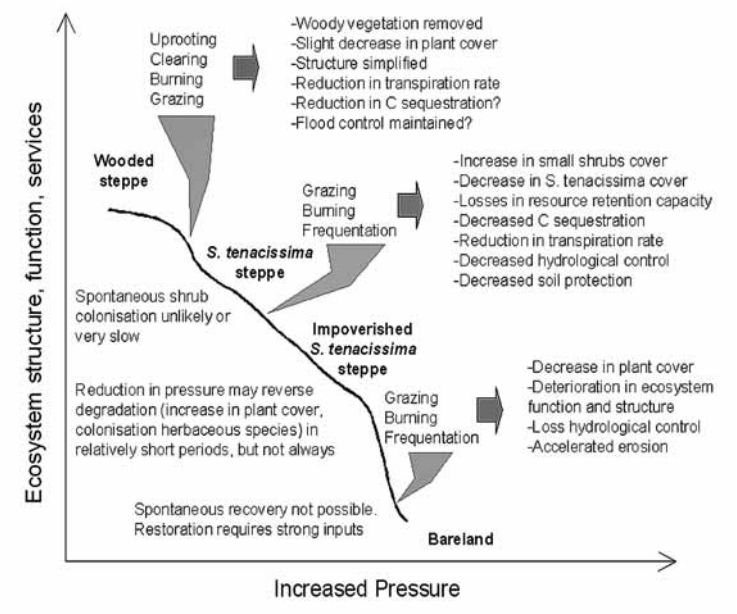


FIGURE 2. A schematic view of *S. tenacissima* degradation and aggradation dynamics in response to external anthropogenic pressures. Direction and intensity of changes in ecosystem structure, function and services will be highly dependent on climate, topography, lithology, and on the variable measured (see comments on the right of the figure). Factors favouring the shift between alternative states, and structural and functional changes associated with them, are indicated for each transition between alternative states. The probability of spontaneous recovery and need for restoration are described on the lower left corner. See further details in the text.

Restoration of *S. tenacissima* steppes should focus on reversing the effects of degradation thresholds by improving soil conditions, increasing plant cover, and introducing key-stone woody species. These could be sequential steps in a long-term (and well funded, well-coordinated) restoration programme. But restoration objectives are usually more modest, involving single steps departing from a given degradation status (i.e. introducing sprouting shrubs and trees in steppes showing high cover of *S. tenacissima*, increasing the cover of *S. tenacissima* and other palatable plants when these species have lost dominance, etc.). In addition, restoration is basically a society-driven process, and thus subjected to society needs and aims. It is not possible to define a single objective for restoring *S. tenacissima* steppes, but a range of context-driven restoration targets. These may emphasize different aspects of ecosystem structure and function, goods and services, including the presence of woody plants, high biodiversity, hydrological control, reduction in fire risk, and improved grazing quality, among others.

In this context, it is possible to explain apparent paradoxes such as the use of woody exotic species in North African *S. tenacissima* steppes, a practice which is generally considered 'taboo' in southern Europe. In North Africa, species such as alien *Acacia* spp. are valued for the production of fuelwood and forage, and soil amelioration, whereas in European steppes, demands for forage and fiber vanished during the second half of the 20th century. In these areas, restoration now focuses on preserving ecosystem function while incorporating species whose abundance was reduced by centuries of intense human use, such as *Pinus halepensis*, *Pistacia lentiscus*, *Quercus coccifera* and *Rhamnus lycioides*. As discusssed above, these are key-stone species that determine the structure and functioning of *S. tenacissima* steppes.

By contrast to degradation thresholds, the concepts of restorability and restoration thresholds have received much less attention. Restorability has been defined as the effort needed to bring the ecosystem to a desired state or restoration target (Zedler and Callaway 1999). 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 persistance (Montalvo *et al.* 1997 *inter alia*).

Concerning restorability, we may ask whether degradation status and restoration effort are directly related. This is obvious when contrasts in ecosystem status are evident, and may be applicable in most cases (Milton et al. 2003). But the relationship is not always so evident. For example Maestre et al. (2006b) compared restoration success, based on the performance of a planted key-stone species, and degradation status in a range of *S. tenacissima* steppes showing contrasted functionality. In this case, environmental factors were more strongly related to restoration success than ecosystem functionality. In addition, plantation success was negatively related to altitude, plant cover, species richness, shrub cover, and water infiltration. Evidence of facilitation between *S. tenacissima* and seedlings of woody species (see below) suggests that competition was not the main responsible for these results.

It has been suggested that the restoration of semiarid *S. tenacissima* steppes should follow a two-step approach according to their functional status and structural attributes (Maestre and Cortina 2004a). In steppes showing clear symptoms of impaired functionality, restoration actions should focus on repairing soil stability, infiltration and nutrient cycling. This can be achieved by using low-cost methods such as the creation of new patches using dead branches (Ludwig and Tongway 1996; Tongway and Ludwig 1996). In steppes with better functional status, restoration actions should focus on the introduction of late-successional shrubs as a way to improve ecosystem functions, to increase ecosystem resilience against disturbances, and to foster the establishment of other plant and animal species. Our results suggest that the first step may not be needed in order to achieve the second, and that shrubs can be established in steppes with reduced functionality and with clear symptoms of degradation. Further exceptions to the direct relationship between ecosystem functioning and restorability include cases where restoration towards a highly functional target ecosystem must be triggered by a major disturbance such as fire or clearing (i.e. by temporary decreasing functionality).

20th century approaches to the restoration of Stipa tenacissima steppes

As previously discussed, restoration priorities depend on the ecological, socio-economical and cultural context. Thus, it is not surprising that restoration objectives and the techniques to achieve them, have changed throughout the 20th century. Restoration of semi-arid steppes in SE Spain, historically focused on hydrological control (e.g. Sierra Espuña, Murcia 1886; see chapter 5, this volume), and dune fixation (e.g., Guardamar dune field; Mira 1906). These were large-scale projects, taking several decades to be accomplished, that involved huge amounts of work-force, building of new infrastructures, and the development of novel practices. These projects used a wide range of techniques and species, and made early attempts to incorporate adaptive management techniques. Some of them now represent outstanding examples of restoration practitioners contribution to human welfare.

Some decades later, pritorities shifted towards employment generation, and the establishment of a forest cover, in addition to hydrological control (Peñuelas and Ocaña 1996). In Spain, an ambitious afforestation programme was launched in 1939. By the end of the programme, in 1986, more than 3.6 milion hectares had been planted, mostly with conifers. *Pinus halepensis* plantations carried out during the second half of the 20th century contribute to the vast area now covered by this species in SE Spain (Vélez 1986), and other Mediterranean areas (Barbéro *et al.* 1998; Ginsberg 2006). Heavy machinery and intense site preparation gradually became part of regular operations with the aim of fostering pine establishment, even if that meant modifying the hydrology of whole catchments. Following prevailing ecological theories on succession and competition, standing plants were considered as potential competitors for planted seedlings and frequently removed.

Under semi-arid conditions, and particularly under stressful conditions, such as on sunny slopes and thin soils, pine plantations have performed poorly. Several decades after plantation, pines show scant cover due to low survival, slow growth rates and pest attacks (Maestre and Cortina 2004c); recruitment is low, and pines have not facilitated the establishment of key-stone sprouting shrubs as originally planned. Despìte that a rigorous evaluation programme has not been implemented to date, there is a general consensus on the need to manage degraded pine plantations to increase biodiversity, foster resource retention and carbon sequestration capacity, and promote ecosystem resilience (Maestre and Cortina 2004c).

As previously discussed, targets for the restoration of degraded pine plantations are diverse, and they should be defined on the basis of plantation status and social needs. Under the best scenario, we may foresee a spatially heterogenous steppe, with *Pinus halepensis* and sprouting shrub patches, in a matrix dominated by herbaceous species (e.g. *S. tenacissima* and *Brachypodium retusum*), and open areas (Fig. 3). In degraded pine plantations, this scenario may be attained by planting sprouting shrubs. These may be preferentially located in open areas, to avoid negative interactions with pines and accompanying herbaceous vegetation. Additional techniques may be needed to ensure establishment

(see below). In addition, canopy opening and seedling tending may be needed to ensure *P. halepensis* recruitment. Finally, herbaceous populations may be reinforced to reach maximum plant cover according to site potential, and minimize interpatch distance and the risk of resource leakage (Tongway and Hindley, 1995; Maestre and Cortina, 2004a).



FIGURE 3. Virtual recreation of expected changes after the restoration of an impoverished *S. tenacissima* steppe in Venta Lanuza, SE Spain. Note the increase in plant cover and the presence of additional woody vegetation, established on favorable sites (ravines, N facing slopes).

Landscape structure, functional state and ecosystem restorability

Restoration should be based on an evaluation of present undesirable conditions. This is an easy task when degradation is extreme. But it is not so when disturbances have not suppressed all vestiges of earlier ecosystems. These are relevant for restoration practitioners for several reasons. First, evaluation may be the first step towards diagnosis and cure. By identifying those aspects of ecosystem structure and function that work well in degraded ecosystems, and those in need to be fixed, the efficiency of restoration programmes can be improved. Very often (but not always) there is no need to further alter the whole ecosystem to initiate restoration. On the contrary, by preserving remaining structure and function in degraded ecosystems, restoration practitioners can make use of them to improve restoration success. For example, if hydrological control of the area to be restored is not bad, it is probably unadvisable –and expensive, to manipulate the hydrology of the whole catchment, a lesson that has been learnt after many failures in Mediterranean drylands (Maestre and Cortina 2004c). Second, evaluation allows the identification of priority areas for restoration. Finally, ecological interactions in degraded ecosystems may be used to improve restoration success. The case of facilitation by extant vegetation is well known in drylands, including *S. tenacissima* steppes (see below).

It is surprising that dignosis and prognosis, procedures that are well established in medical care, economy, and other areas such as artistic and archaeological restoration, have received much less attention in ecological restoration. It should be considered as unreasonable to implement a large scale plantation in a degraded area that has not been previously and carefully evaluated, as prescribing a major medical intervention on the sole basis of expert judgement. Unfortunately, this situation is not uncommon. The restoration of *S. tenacissima* steppes has been frequently based on the plantation of tree species, after poor evaluation, disregarding existing ecosystem structure and function.

Several methods have been developed for the evaluation of ecosystem structure and function in arid lands (Tongway and Hindley 1995, Herrick et al. 2005). These assessment protocols are designed for being implemented at a management scale. The method developed by David Tongway and colleagues at CSIRO (Landscape Function Analysis, LFA) is based on an evaluation of the spatial structure of resource sources and sinks in open plots, and on a semiquantitative evaluation of surface soil properties. This method estimates three landscape function indices that are related to water infiltration, soil surface stability and nutrient recycling, respectively. In *S. tenacissima* steppes, LFA indices are related to structural variables such as distance between consecutive resource sinks, cover of sprouting shrubs, and species richness (Maestre and Cortina 2004a). *Stipa tenacissima* tussocks commonly show high values of the three indices (Maestre and Cortina 2004a), reflecting their capacity to concentrate resources.

The use of biological soil crusts

As mentioned above, BSC are a prominent feature of arid and semi-arid ecosystems, and are very common in *S. tenacissima* steppes. Albeit the effects of BSC on the functioning of *S. tenacissima* steppes have only begun to be explored, available evidence suggests that they can play key roles in the source-sink dynamics of water and sediments (Maestre *et al.* 2002a; Martín *et al.* 2003), and the establishment of vascular plants (Navarro-Cano *et al.* 2003), but their capacity to fix nitrogen, increase soil organic matter content, and affect *S. tenacissima* performance is rather limited (N. Martín, unpublished data). Given their critical role in ecosystem function, and the increasing awareness on their importance as a key component of natural ecosystems, it is not surprising that there is renewed interest in the response of BSC to anthropogenic and natural disturbances (Belnap and Eldridge 2001). These crusts are very sensitive to disturbances, and temporal estimates for their recovery under natural conditions typically are in the range of decades to millennia (Belnap and Eldridge 2001). To overcome this limitation, and to speed up recovery, *in situ* inoculation of soils with biological crusts components, such as cyanobacteria, has been recommended in degraded arid and semiarid ecosystems (Belnap 1993, Buttars *et al.* 1998). In *S. tenacissima* steppes, cyanobacteria colonization can be promoted by applying a mix of BSCs and water, together with irrigation and organic soil amendments (Maestre *et al.* 2006a).

Further studies are needed to increase our knowledge on the ecology of BSC in *S. tenacissima* steppes. These could be complemented with studies devoted to develop suitable application techniques at a management scale, and to isolate native cyanobacteria for *ex situ* mass culturing methods (Buttars *et al.* 1998). Such a development would minimize the collection of intact biological crusts from undisturbed areas to obtain inoculum, one of the main drawbacks of using inoculation techniques to restore biological soil crusts in degraded areas.

Facilitation by Stipa tenacissima as an aid in restoration

The succesful establishment of vegetation during the restoration of semi-arid ecosystems is a challenging task due to the harsh climatic conditions, to the low soil resource levels, and to the scarce and unpredictable rainfall regimes that characterise these environments (Whisenant 1999). Important research efforts have been devoted in the last decades to overcome these limitations, and nowadays there are numerous management techniques to improve plant establishment during restoration of arid and semi-arid environments. Among these techniques, the use of positive interactions among neighbouring plants is especially appealing. While facilitation has been documented in a wide variety of environments (Callaway 1995), it is by far most common in arid and semi-arid ecosystems (Flores and Jurado 2003). The use of facilitation in restoration would allow the use of the remaining structure and functioning of degraded ecosystems into management, an issue as largely advocated by ecologists as rarely employed in practice (e.g. Wallace et al. 1980). Despite the a priori potential and attractiveness of facilitation as a restoration tool, and the large number of studies emphasizing its importance as a driver of community structure and ecosystem dynamics, it has been largely neglected in the restoration programmes carried out in semi-arid areas for decades.

As discussed above, *S. tenacissima* tussocks accumulate resources, and they have been found to facilitate the establishment of BSC and vascular plants. In order to evaluate the potential of facilitation to improve the restoration of *S. tenacissima* steppes, we conducted a series of experimental plantings in steppes located in the province of Alicante (SE Spain; Table 3). In these experiments, we introduced seedlings of different shrub species under the canopy of *S. tenacissima* tussocks and in bare ground areas devoid of vascular plants. The results obtained were mainly dependent on the climatic conditions of the first year after plantation, the species considered and the presence of *S. tenacissima*. This species facilitated the establishment of the introduced seedlings in most cases where mortality was not complete. However, as discused above, the effect was not universal, and negative interactions between *S. tenacissima* and the introduced seedlings were observed under high abiotic stress.

These results are not surprising. Neighbours may increase water availability if shading reduces evaporation (Maestre et al. 2003a), and by improving soil properties like texture and soil organic matter (Puigdefábregas *et al.* 1999), and may reduce it through direct water

TABLE 3. Results of experimental plantings evaluating the effect of *Stipa tenacissima* on the survival of one-year-old seedlings of Mediterranean woody shrubs. In all cases, the seedlings were planted using hand-made 25 x 25 x 25 cm planting holes. YE = Planting year, SP = Species, SI = Name of the experimental site, ST = survival of seedlings planted in the vicinity of *S. tenacissima* tussocks (in %), SO = survival of seedlings planted in open ground areas devoid of vascular plants (in %), DU = duration of the study (in months), RA = rainfall accumulated during the first year after planting (mm), SO = source of data.

YE	SP	SI	ST	SO	DU	RA	SO
1998	*Quercus coccifera*	Aguas	5	7	12	212	Maestre *et al.* (2001)
		Ballestera	13	4	12	132	
		Campello	20	2	12	197	
	Pistacia lentiscus	Aguas	10	3	12	212	
		Ballestera	16	15	12	132	
		Campello	17	9	12	197	
	Medicago arborea	Aguas	85	78	12	212	
		Ballestera	69	30	12	132	
		Campello	85	77	12	197	
1999	*Quercus coccifera*	Aguas	0	0	12	264	Maestre *et al.* (2002b)
		Ballestera	0	0	12	150	
		Campello	0	0	12	193	
	*Quercus coccifera**	Aguas	0	0	12	264	
		Ballestera	0	0	12	150	
		Campello	0	0	12	193	
1999	*Quercus coccifera*	Aguas	0	0	24	264	Maestre (2002)
		Ballestera	0	0	24	150	
		Campello	0	0	24	193	
	Pistacia lentiscus	Aguas	8	6	24	264	
		Ballestera	0	3	24	150	
		Campello	3	0	24	193	
2001	*Pistacia lentiscus*	Aguas	57	32	15	225	Maestre *et al.* (2003)
		Ballestera	4	0	15	149	
2003	*Pistacia lentiscus*†	Albatera	0	0	17	133	Maestre et al. (2006b)
		Jijona	72	72	17	125	
		Lanuza	56	23	17	134	
		Marquesa	83	67	17	156	
		Finestrat	40	60	17	171	
		Fontcalent	47	88	17	109	
		Palomaret	13	0	17	139	
		Peñarrubia	0	0	17	187	
		Relleu	89	65	17	150	
		Ventós	0	0	17	102	

* Seedlings inoculated with sporal inoculum of *Pisolithus tinctorius* in the nursery.

† Rainfall values correspond here to the first eight months after planting.

uptake and rainfall interception (Valladares and Pearcy 2002; Bellot *et al.* 2004). We argue that, in strongly water-limited environments, we should expect facilitation only when neighbours increase availability beyond their own water uptake requirements, allowing increased benefits in terms of improved soil fertility and microclimate to increase in plant performance compared to areas without neighbours. We suggest that a threshold level in water availability will define the transition from net negative to net positive interactions (Cortina and Vallejo 2004, Maestre et al. 2005b).

These experiments show that facilitation has potential to improve the restoration of *S. tenacissima* steppes, specially under conditions of moderate abiotic stress. It must be noted, however, that rainfall is an overriding factor in determining the success of the experimental plantations conducted in semi-arid environments (see Cortina et al. 2004 for a review), and that competitive effects are expected under extreme abiotic stress conditions. Improvement in our ability to forecast rainfall is thus critical in order to effectively use facilitation by *S. tenacissima* to improve the restoration of degraded steppes.

Ecotechnology as a replacement for ecological interactions

Restoration practice must be based on a comprehensive understanding of abiotic and biotic drivers of ecosystem functioning, and careful identification of ecosystem components (Cortina *et al.* 2006). The identification of degradation thresholds is a crucial step in restoration. On the one hand, they are critical phases in ecosystem degradation and recovery, and thus they can be of great help to select priority areas for restoration. On the other hand, the identification of such thresholds and the interpretation of their underlying causes, may help to recognise which abiotic and biotic ecosystem elements, including disturbances, need to be tackled by restoration programmes. Finally, manipulation of such elements provides excellent opportunities for ecotechnological development, a fact that has probably contributed to the extraordinary success of ecological restoration.

In *S. tenacissima* steppes, there is a large number of ecotechnological tools based on current knowledge on ecological interactions that can be used to improve restoration success (Table 4). Selection of keystone species well adapted to the environmental conditions prevailing in the area under restoration may help restoration practitioners to tackle biotic thresholds. Other options to reverse biotic shortcomings include: (1) the wide range of techniques for controlling unwanted species (including fire, clearcutting, biologic control and the use of biocides), (2) the use of mycorrhizal inoculum, (3) the production of high quality seedlings, and (4) the use of facilitative interactions, among others. Abiotic thresholds may be reversed by locally improving soil conditions (including the application of allogenic soil and soil redistibution), building runoff concentration structures, and using stone pavements and a wide variety of soil amendments, mulches, blankets, etc.

TABLE 4. Ecotechnological tools for the restoration of semi-arid ecosystems. Ecosystem processes in which these techniques are based are briefly described.

Technique	Ecological basis	Reference examples
Biotic thresholds		
Fire, clearing, biocide application, biological control	Control of unwanted species and whole communities, reset succession	D'Antonio and Meyerson (2002), Baeza *et al.* (2003)
Species selection	Biogeographical and evolutive constrains, biodiversity-function relationships	Cortina *et al.* (2004)
Genotype selection	Hybrid vigor, fenotypic plasticity	Alía (chapter 6, this volume)
Field and nursery mycorrhizae and rhizoflora inoculation	Exo and endosimbiotic microflora. Increase in resource availability, protection against pathogens and stress.	Herrera *et al.* (1993), Maestre *et al.* (2002b), Caravaca *et al.* (2003)
Improved seedling quality	Seedling acclimation, avoid early mortality and improve establishment	Vilagrosa *et al.* (2003), Villar (chapter 7, this volume)
Nurse species	Facilitation by reducing stress and consumer's pressure	Maestre *et al.* (2001), Maestre *et al.* (2003b)
Abiotic thresholds		
Branches, silt fences, mulches	Resource sinks, eventually islands of fertility. Runoff concentration together with seeds, sediments and nutrients.	Ludwig and Tongway (1996), Tongway and Ludwig (1996)
Perches	Birds rests. Propagule concentration. Stemflow inputs of water and nutrients, plus shadow)	Verdú and García-Fayos (1996), Pausas *et al.* (in press), Wunderlee (1997)
Organic and inorganic soil amendments	Soil fertility, islands of fertility. Widespread and localised improvement in soil fertility	Valdecantos *et al.* (1996), Valdecantos *et al.* (2002)
Stone piles around planted seedlings	Simulated gravel accumulation on resource links. Shadow plus moisture conservation	–
Stone pavements	Runoff generation towards resource sinks	E. De Simón, pers. com., Hillel (1991)
Treeshelters	Shadow, protection against herbivory. Simulates facilitative interactions.	Bellot *et al.* (2002), Oliet *et al.* (2003)
Soil preparation, including microcatchments, terracing, etc.	Resource sinks, mainly water harvesting.	Whisenant *et al.* (1995), Boeken and Shachak (1994)
Cyanobacteria inoculation	Biological soil crusts. Soil protection, alteration of water and N fluxes, and vascular plant performance	Buttars *et al.* (1998), Belnap (1993), Maestre *et al.* (2006a)

Concluding remarks

The further we deepen our knowledge on the composition, function and history of semi-arid ecosystems, the more we recognize their complexity. In contrast to mesic systems, processes occurring at an individual scale, and the spatial arrangement of organisms and resources play a key role in the functioning of semi-arid ecosystems. Moreover, spatial heterogeneity hampers the use of universal recipies. Ignoring these features has been the cause of past failures of restoration programmes, and social disappointment. Fortunately, ecological knowledge is being gradually incorporated into management practices in this area, as revealed by a change in the magnitude and degree of accuracy of restoration programmes, an amelioration of restoration tools and practices, and a diversification of restoration objectives. Some limitations associated to complexity and history still represent major challenges for the restoration of semi-arid Mediterranean ecosystems. For example, reference ecosystems are not easily identified in these areas because records of pristine ecosystem are lost, because there may be not one but several potential alternatives, and because environmental conditions may have substantially changed and may further change in the future. Thus, in semi-arid steppes, restoration should focus on short-term achievements, and desirable ranges of ecosystem structure and function (and the associated natural capital, Aronson et al. 2006), rather than aiming at the recovery of an ancestral community (Cortina *et al.* 2006).

Processes in semi-arid areas are slow, and the temporal scale for implementation and evaluation of restoration actions should be adjusted accordingly. In addition, intra- and inter-annual climatic variability are high. This adds further complexity to the evaluation of restoration actions. As a result of such complexity, our knowledge on long-term ecosystem dynamics, including a thorough understanding of the drivers of successional trajectories and the way they interact with climatic variability, is still very poor. This has critical implications on the way we plan and evaluate restoration in semi-arid areas, and emphasize the importance of implementing adaptive management techniques.

Climatic trends deserve further attention. Predictions for southern Europe suggest that the whole area will get warmer and drier (De Castro *et al.* 2004). This will affect restoration practices in many ways. On the one hand, ecosystems such as *S. tenacissima* steppes, may be particularly sensitive to climatic changes. Substantial alterations of *S. tenacissima* steppes structure and function may occur in the next future, particularly in the limits of current geographic distribution, and in areas subjected to other sources of stress, such as overgrazing. There are strong evidences of such abrupt changes in *S. tenacissima* steppes (Aïdoud and Touffet 1996). On the other hand, the outcomes of restoration are strongly dependent on climatic conditions. Techniques to attenuate the effects of dry spells may be needed to foster seedling establishment, including watering, a practice that is uncommon in SE Spain, but regularly used in N Africa. As has been previously mentioned, fast climatic changes represent a major challenge for ecological restoration. Will *S. tenacissima* steppes be sustainable under a warmer and drier climate?. What kind of ecosystem should we promote in this area if the temporal scale for cli-

matic changes is apparently similar to or even faster than that of ecosystem dynamics? Should we persist in avoiding the use of alien species, or we should rather use those species that may ensure a particular degree of ecosystem function, independently of their origin?. Providing proper answers to these questions represent a major challenge for researchers and practitioners involved in the restoration of *S. tenacissima* steppes.

Finally, vast areas covered by *S. tenacissima* steppes are being transformed into urban developments and trivial landscape. As an example, in the Region of Valencia the surface area covered by buildings grew by almost 50% between 1990 and 2000 (Greenpeace 2005). Similar examples are found in both rims of the Mediterranean. Emphasis on urbanisation suggests that functional and diverse ecosystems do not currently represent a major social priority, at least for local decision-makers. On the other hand, restoration projects are being currently implemented by urban developers to provide additional aesthetic and economic value to built areas. Moreover, NGO's and privates are gradually getting involved in small-scale nature conservation and restoration projects in response to high rates of land consumption. Both types of initiatives represent new ways of implementing restoration, and novel alternatives to traditionally centralised restoration funding schemes. At this point, however, it is worth to remember that restoration must always be a second alternative to conservation.

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