Land Restoration to Combat Desertification

Innovative Approaches, Quality Control and Project Evaluation



Edited by
Susana Bautista
James Aronson
V. Ramón Vallejo

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Preface

Many drylands in the world suffer problems of land degradation and desertification derived from human activities and exacerbated by drought. Too often these degradation processes have been endured by the ecosystems for a long time, and, according to forecasts of climate change, are likely to worsen in the future. Ecological restoration combined with adaptive management can be effective tools in response to this environmental and socioeconomic problem.

Reforestation and afforestation are restoration actions traditionally used to recover degraded lands for production and to alleviate on-going degradation processes. In some cases barren land has yielded magnificent forests, and in other cases the impacts are less clear. Despite the long-standing experience among scientists and land managers in reforesting degraded lands worldwide, in general, assessments of the results of land restoration projects are limited either in terms of data or breadth of perspective, and therefore little of real use can be drawn from this work. The lack of available scientific and technical information on restoration actions hampers the dissemination of technology within and among countries and regions, and the sharing and more comprehensive application of the best technology and approaches available. The need for more systematic and comparable evaluation of the results of restoration and management actions and more effective knowledge exchange and dissemination is widely acknowledged. The information needed includes biological, ecological and socio-economic aspects.

Recently, a number of different research and development projects funded by the European Commission have promoted the development of innovations in dryland restoration and reforestation. To improve scientific and technical communication on land restoration, and to capitalise on recent scientific advances in this area, the REACTION project (Restoration Actions to Combat Desertification in the Northern Mediterranean, www.ceam.es/reaction) was launched under the Fifth Research, Technology and Development Framework Programme of the European Commission, in the key action area "Climate Change and Ecosystems". The project has led to the establishment of a Northern

Mediterranean network, information system, and database on land restoration to fight desertification, and the development and testing of an indicator-based protocol to evaluate the results of forest restoration projects in the Mediterranean. The present book summarizes the main achievements of the REACTION programme and provides a series of restoration guidelines developed in the light of past and present innovative approaches.

The objective of this book is to present the latest scientific and technical advances in land restoration with the purpose of combating desertification in arid, semi-arid and dry-subhumid regions, with emphasis on reforestation actions in the Mediterranean region. This includes the identification of the innovative aspects of the land restoration process, from project planning to execution and the monitoring of results, as well as restoration technology, from plant production in the nursery to planting or seeding. Specific attention is paid to the discussion and development of criteria and procedures for quality control of all processes forming part of forest restoration projects, from seed collection to monitoring protocols. As a further outcome of REACTION, we also present a methodology for long term forest restoration assessment on the basis of updated, practical information of particular interest to practitioners in dryland restoration, e.g., in the framework of the United Nations Convention to Combat Desertification, and the respective National and Regional Action Plans. We do not attempt to address all possible, and especially theoretical aspects of ecological restoration, but only those we consider at the cutting edge of practical land restoration. We hope it will help.

Susana Bautista, James Aronson and V. Ramón Vallejo

I.

Introduction

Problems and Perspectives of Dryland Restoration

1

V. Ramón Vallejo

Introduction

Forest degradation was probably necessary for human cultural evolution, especially for the development of agriculture and animal husbandry. Quite early, however, our ancestors realised the essential asymmetry underlying deforestation and desertification. It is easy to destroy forests, but their recovery is agonizingly slow, if they recover at all. This is nowhere more evident than in the Mediterranean and circum-Mediterranean lands, where some of the first human experiments in transformation of land, urbanization and stable governments were carried out. As early as the fourth century BC, Plato eloquently described widespread and profound human impacts on forests: *"Hills that were once covered by forests and produced abundant pasture now produce only food for bees"*. The degradation-regeneration asymmetry referred to just now is caused by the increased entropy of ecosystems related to the loss of organisation that had evolved over long periods of time. Therefore, while human or non-human disturbances may cause sudden ecosystem (forest) degradation, repair and reconstruction require a long time (fast out, slow in, for example in organic carbon budgets). When positive feedback processes of degradation appear after disturbance, leading to irreversible loss of ecosystem integrity, artificial inputs of energy are needed to stop and reverse degradation.

The recognition of the need for forest restoration is also far from new. Attempts to reforest degraded lands are documented as far back as the Middle Ages (Manuel Valdés and Gil 1998), and deliberate introduction of non-native forest species is known from much earlier times, especially during the Roman empire. Throughout the 18th – early 19th centuries, the so-called Age of Enlightenment, European administrators attempted to preserve and promote forests. However, only since the late-19th century did afforestation efforts attain significance at national scales, finally becoming widespread – if not yet fully developed – during the 20th century.

During the first half of the 20th century, many large afforestation and reforestation projects were conducted in the Mediterranean region, and elsewhere. As an example, the

Spanish National Reforestation Plan, initiated in 1939, led to tree planting – mostly pines – on more than 4 million hectares, in the course of a century (Fig. 1). The results of these efforts were diverse, but nowadays we are able to enjoy, and benefit from, many magnificent forests on formerly degraded land as a consequence of these efforts (Vallejo 2005). Similar efforts were made in France (Vallauri et al. 2002), Portugal (Campos Andrada 1982), Italy (Hall 2005) and further east and south as well. In the last quarter of the 20th century, huge advances were made in the field, both conceptually or technically, as part of the wider, indeed global, movement to begin investing more time, energy and financial capital into the restoration and rehabilitation of degraded and ill-used ecosystems on which all human economies are ultimately dependent. In the 21st century, new challenges – notably climate change, and ongoing human population growth, will require new strategies. We will address these issues at the end of this chapter

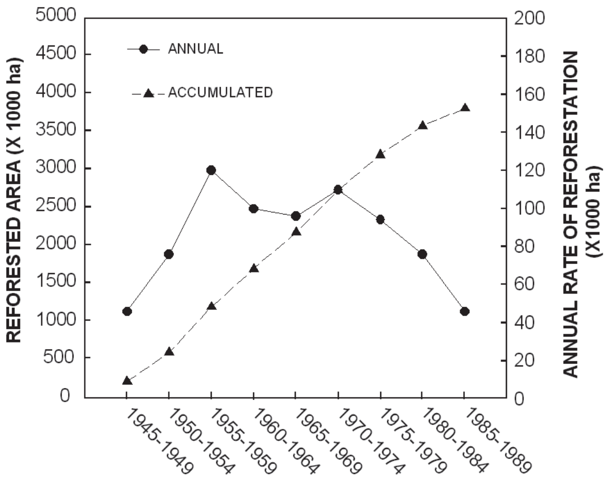

FIGURE 1. Spanish National Reforestation Plan (approved in 1939) (Ortuño 1990).

Here, and throughout this book, we address wildland restoration in a wide sense. In the Mediterranean Basin, as in other seasonally dry regions, the distinction between forests and shrublands is not straightforward. There are frequent and subtle transitions among open woodlands, low forests (sometimes coppices) and tall shrublands (*maquis, garrigue, etc.*), perhaps especially at the drier end of the range of Mediterranean bioclimatic conditions (Aronson et al. 2002). In addition, many sclerophyllous shrubs may attain tree-size if allowed enough time and space to grow. In ancient Roman times, *silva* (forest) was clearly distinguished from *saltus* (uncropped forest region and other wildlands used for grazing). Under the Spanish legislation, however, forest lands include all wildlands under the term *monte*, even those without continuous tree cover. For the Inter-governmental Panel on Climate Change (IPCC), in its efforts to establish carbon budget accounting, "forest" is defined in very broad terms as a minimum area of 0.05-1.0 ha with tree crown cover of more than 10-30%, including trees with the potential to reach a minimum height of 2-5 m at maturity (UNFCCC 2002), also including young stands resulting from plantations or regenerating after disturbance. According to that definition, a great number of plant formations of arid lands – including open woodlands and tall shrublands, could be considered as forests under the Kyoto Protocol. Considering these peculiarities of the drylands, hereafter we will use land and forest restoration in the wide sense of assisting the recovery of degraded lands as per the SER International's Primer of Ecological Restoration (SER 2002) towards any natural wildland type, such as shrubland, woodland and forest *sensu stricto*.

Arid and semi-arid lands are highly sensitive when faced with anthropogenic forces of perturbation and degradation. Plant cover is scarce and especially vulnerable to disturbances, unpredictable and prolonged periods of drought. In addition, plant recovery after damage is very difficult under these stressful conditions, and this applies both to natural regeneration and to artificial restoration. Therefore, as plant regeneration is the major driver of ecosystem recovery, land restoration is especially necessary and, at the same time, difficult in arid and semi-arid lands.

Major difficulties of reintroducing plants in Mediterranean degraded lands are related to a combination of high water stress and regimes of high risk of varied post-plantation disturbances, including fire and grazing by domestic or wild animals, combined with relatively moderate impacts from competition (Fig. 2). In moister environments, stress is lower and competition higher. Therefore, restoration of Mediterranean ecosystems would especially require addressing ways of overcoming water stress and avoiding or mitigating the impact of disturbances.

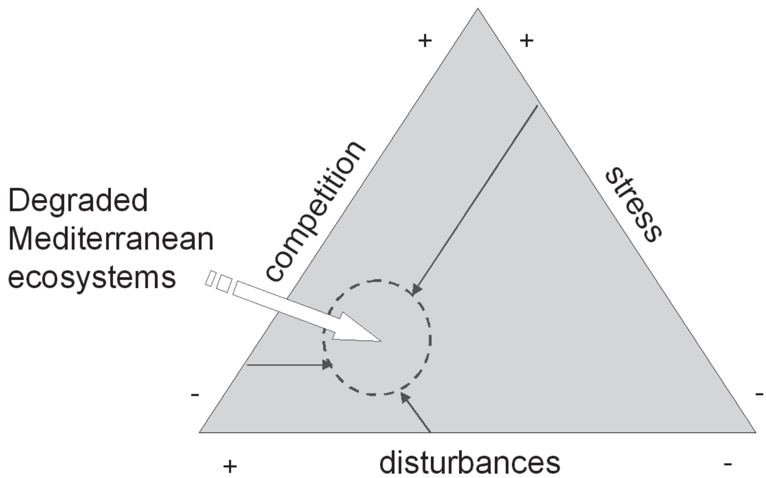

FIGURE 2. Main limitations for reintroducing plants under Mediterranean conditions. Triangle inspired by Grime's work on plant life history strategies (1979).

Old *vs.* new approaches

Traditional reforestation projects conducted in the Mediterranean countries were not, strictly speaking, ecological restoration projects as we understand this term nowadays. However, they embraced the broad aims of restoration, such as reducing soil erosion and runoff, or recovering natural forests, though sometimes exotic species were used as intermediate stages in the rehabilitation process (e.g., Vallauri et al. 2000, Hall 2005). In Mediterranean countries, most afforestation projects addressed watershed protection, reducing floods and soil erosion, and, in coastal areas, stabilization of mobile dunes. In addition, a common goal was to increase forest surface and productivity, while improving rural economies through the creation of jobs and livelihood opportunities provided by the project execution investments and the expected increase in timber and non-timber forest productivity.

Strong socio-economic development in southern European countries during the second half of the 20th century lead to profound changes, with rapid and dramatic depopulation of many rural areas, changes in land use – agricultural land abandonment, reduction of grazing and firewood collection, etc. – and a decreasing dependence of rural people on forest resources. Slowly, a new perception of nature was growing in the EuroMediterranean countries and elsewhere that lead to new demands on wildlands, leaning more towards recreation, and ecological, cultural and landscape values. Of course, these new demands required, and still require, a corresponding adaptation of forest restoration techniques to meet these demands. The transition between old and new objectives is best characterised by a shift from reforestation, tree-oriented interventions based on the planting or seeding of both native and non-native trees, followed by silviculture, to ecosystem-oriented, ecological restoration of native ecosystems within vibrant cultural landscapes. This requires the diversification of plant species used in restoration projects, and making better use of the large pool of native species available, as well as considering fauna, microorganisms and soils along with plants.

The use of larger number of native species in particular requires the corresponding research and development on their autoecology, ecophysiology and appropriate cultural techniques. Accordingly, the recent afforestation measures for setting aside agricultural lands, promoted under the Common Agricultural Policy of the European Union, was conceived to aid in recovering native forest ecosystems. In the future, much more attention should be paid to the “hidden” part of ecosystems, particularly soil microorganisms that play such a critical role in the restoration process.

At present, differences in socio-economic development, degree of dependence on natural resources among countries around the Mediterranean, and among regions within countries, make possible the more or less comfortable coexistence of old and new approaches and techniques in afforestation. The challenge is to harmonize restoration strategies and techniques for sustainable, multipurpose afforestation/reforestation.

Local vs global

In the current era of globalisation, we widely assume the principle of thinking globally and acting locally. And this especially holds for land restoration. In science, of course, we must try to derive principles of general application. But land degradation, and the co-related land restoration, are the consequence of specific impact of local actions on local ecosystems along specific time frames. Therefore, the multiple combinations and outputs of this set of specific variables make it difficult to derive generic protocols or solutions for ecological restoration. Therefore, we recommend developing specific restoration strategies for specific regions with a shared history and biophysical setting. In this book, we focus on the Mediterranean Basin, and secondarily, on other seasonally dry lands elsewhere, assuming that transferring the guidelines to other biogeographic and socio-economic contexts will require the reformulation of approaches. However, acting locally can, to some extent, have global consequences. Local actions, such as restoration projects, if developed at sufficiently large scale, may trigger feedback processes that could influence local climate and ultimately even at the global scale, and this might be especially the case in the Mediterranean Basin (Millán et al. 2005, Clewell and Aronson 2006, 2007; see Chapter 2, this volume).

Structural mismatching of forest management policies

It is widely accepted that land and forest management must respond to social demands, and this includes forest restoration (Lamb and Gilmour 2003; see Chapter 3, this volume). The structural problem of the forest sector in industrial and post-industrial societies is that social demands, and their expression in forest policies, change faster than forest ecosystems grow and develop. Consequently, forest policies that respond to current demands from forests (or more generally from land use interests) may become obsolete in only a few decades, leaving to future generations a problem that may be difficult to reverse, or definitively irreversible. Examples of this time mismatching are: a) the clearcutting of cork oak woodlands conducted in Portugal for wheat production during the 1930s (Roxo et al. 1999), the later abandonment of many of these fields because of poor soil productivity, and the recent attempts to recover cork oak in these now degraded soils; and b) the eucalyptus plantations established in dry areas of western Spain in the 1960s, which are now abandoned, suffering frequent wildfires and, in some cases, uprooted at a large economic cost to restore the native forest. To overcome these contradictions, land use and forest management policies should have long-term perspectives, keeping ecosystems under the threshold of reversibility for any other future productive use. This is a basic assumption of sustainable and prudent land use: potentially productive lands should at all costs be protected and buffered outside of political fluctuations.

Evaluation of restoration efforts

Natural values form part of the icons of welfare in post-industrial societies, including degraded land restoration. This assumption allows and encourages widely accepted investment of public and private funds in land restoration. However, land restoration is still in a kind of early development stage where public valorisation criteria mostly rely on rough quantities (e.g., afforested surface area) with little, if any, evaluation of the quality (and persistence) of that restoration effort. Afforested/reforested acreage is easily confronted in political discussions with fire-damaged acreage as indicator of environmental policy efficiency, easily convertible in budgetary terms. Evaluating at this simple level may promote restoration actions *per se*, without sufficient justification, and hinder the prioritisation of really necessary and good quality projects. Quality control and detailed, scientifically based evaluation of restoration projects are essential elements of their performance (as in many other economic activities) that would contribute to the optimisation of restoration investments and delivery of feedback from restoration experiences into the improvement of the processes. In this spirit, the EU-funded REACTION project was launched with the primary goal of developing and propagating tools for evaluating forest restoration projects in the Mediterranean region. Quality control and evaluation of restoration projects is a major subject of this book, especially in Chapters 2 to 5.

The evaluation of restoration projects should consider ecological, technical, and socio-economic issues (see Chapter 2, this volume, and Clewell and Aronson 2006). Innovation in restoration technology often derives from sophisticated techniques that highly increase implementation costs. However, such technical developments should be accompanied by careful cost-benefit analysis so as to avoid extreme economical unsustainability (see below). Indirect, passive restoration techniques usually are much cheaper than direct interventions and accordingly they deserve greatly increased attention.

The challenges of the near future: the perspectives of climate and land use change

Mitigation vs. adaptation to climate change

Future perspectives of forests and land restoration need to be considered in the perspective of land use dynamics, primarily driven by human demands on goods and services. Recently, Rounswell et al. (2006) have proposed future land use scenarios for Europe on the basis of the IPCC emissions scenarios (Nakićenović et al. 2000). Scenario changes are coincident in a generalised increase of agricultural land abandonment. This would reinforce present trends in Southern Europe with regard to the spread of wildfires, the increase of desertification processes, and more opportunities for restoring degraded lands.

Apart from the outstanding and historical role of land use change as a driver, a newly recognised process will undoubtedly have great influence on land management and restoration strategies, i.e., anthropogenic climate change. Now and in the future, global society should incorporate in the formulations of land restoration the limitations (and opportunities?) of climate change, or better the risks associated to climate change projections (Harris et al. 2006). For the Mediterranean, these projections foresee an increase of drought intensity and frequency of extreme events (drought spills, heavy rainstorms) and induced disturbances such as wildfires and flash floods for the next century or so (IPCC 2001), that is in the time window of full development of restoration projects that are initiated right now. The increase of water shortage in the Mediterranean would be especially acute in transition regions that may trigger dramatic changes in ecosystem composition and structure, e.g., from dry sub-humid to semi-arid (loss of forest cover potential), and from semi-arid to arid (strong reduction of plant cover). However, the relative effects of land-use induced degradation and climate-change induced impacts, and their interactions, are difficult to advance so as to incorporate them in the restoration practise. Of course, the combined effects of human disturbances and climate change are far from linear. Soil properties are very sensitive to land use impacts, and only slowly responsive to climate change. In fact, relict soils having supported various climate change cycles are frequent in the Mediterranean and other warm regions in the world. We can hypothesise that in extremely disturbed sites, climate change impact would be comparatively lower than in less disturbed sites. In view of these perspectives, the key question is how to design restoration projects, and what references should be used (Fig. 3). Should we accommodate our techniques and species selection to the expected aridisation of the climate, that is taking a pure adaptation approach, or should we use species and techniques trying to mitigate climate change?

According to the IPCC, mitigation is an anthropogenic intervention to reduce the sources or enhance the sinks of greenhouse gases (IPCC 2001). Land restoration would contribute to carbon sequestration, so reducing a major driver of climate change. This area was under discussion in the negotiations of the Kyoto protocol on two headings: afforestation/reforestation and revegetation. Direct human-induced conversions of land through afforestation (on non-forest land in the past

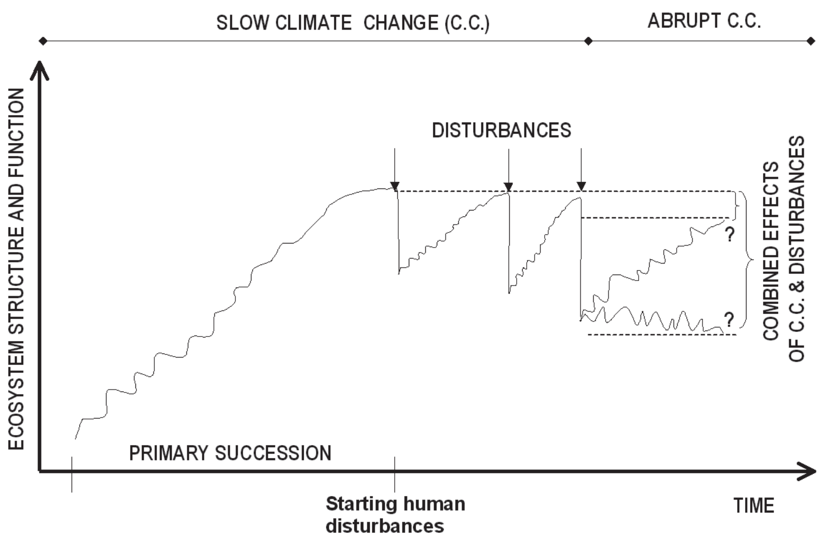

FIGURE 3. Schematic hypothetical changes of ecosystem structure and function over time. There are large uncertainties concerning the combined effects of the two main drivers – direct human disturbances and projected abrupt climate change resulting from anthropogenic modifications to the atmmosphere. How these impacts modify ecosystem structure and processes will have great significance in determining how to choose and utilise reference systems for restoration projects.

50 years), reforestation (on non-forest land for less than 50 years) or revegetation (other vegetation establishment activities) are considered to increase carbon stocks (UFCCC 2002).

Adaptation is defined as the adjustment in natural or human systems in response to actual or expected climatic *stimuli* or their effects, and to degradation drivers and impacts, which moderates harm or exploits beneficial opportunities (IPCC 2001). Adaptive land restoration should adjust landscapes and ecosystems to the expected climate change, including their role as providers of goods and services. At the planning scale, one relevant question would be how to design ecosystems spatial distribution in the landscape to optimise their response to the major foreseen threats in the Mediterranean and other drylands: increased water scarcity and social water demands, protecting watersheds against flash-floods, and reducing wildfire risk. In fact, these threats are already present, though are supposed to increase in the near future. At the ecosystem level, one major question would be to what extent must we anticipate climate change introducing species or ecotypes characteristic of drier regions? Or should we preserve the "original" vegetation as it is currently understood? Harris et al. (2006) point out the difficulties in finding or choosing references under conditions or scenarios of abrupt climate change. In the ACACIA assessment of potential effects of climate change in Europe, Parry (2000) suggested using tree provenance of more southern origin and wider spacing in plantations as adaptation measures. Fully adaptive measures may reduce mitigation effects (e.g., lower capability of sequestering carbon) and the possible feedbacks on local climate. Of course, there are many uncertainties concerning the consequences of these choices and specific research should be conducted to reduce risks of undesired outcomes. One major question to address is how 'restorable' are our extremely degraded ecosystems right now, and at what technological and economic cost could restoration be achieved (see Chapter 8). Answering this question would not only allow greater chances of success in restoration projects but also provide clues on how the foreseen climate change would affect restoration thresholds.

The need of integrating land restoration in the economy: Is carbon sequestration/emissions trading under the Kyoto protocol a viable strategy and tool for restoration?

Land restoration is, in many cases at least, and perhaps especially in developed countries, uneconomical in market economy terms. Restoration works are indeed often expensive or extremely expensive, and most of their direct benefits, such as improving biodiversity and habitat, reducing soil erosion, improving carbon sequestration, improving aesthetic and cultural values of the landscape and so on, are "externalities" – as defined by conventional economic measures. To date, the market deficit is assumed by public and charity funds, and these are unstable and vulnerable to other demands for resources. One promising way of stabilising and rationalising restoration economics would be to introduce the benefits in the marketplace through the economic valuation of goods and services provided by restored ecosystems (Harris and van Diggelen 2006, Rees et al. 2007) and try to include the reparation costs on the degrading agents and/or on the beneficiaries, though in many cases that approach is not possible and financial responsibility falls to the taxpayers (see Holl and Howarth 2000). European Union, Australian, and United States regulations on quarry restoration by the exploiting company are good examples of internalisation of restoration costs by the direct beneficiary of the exploited resource through assurance bonding. In any case, this is a very complex task that deserves much attention and social debate (see Aronson et al. 2007, and Chapter 2, for further discussion, in the context of a new paradigm called restoring natural capital).

Kyoto protocol agreements offer a promising opportunity for funding restoration through the possibility of linking goals of sequestering carbon with restoring degraded lands (so combating desertification (UNCCD) and even improving biodiversity (CBD)). Therefore, bonuses could be transferred from emissions to sequestration through restoration. Along these lines, the European Parliament (Rey and Mahé 2005) has suggested developing market mechanisms for member states for maintaining and increasing carbon sequestration in European forests, through funding forest externalities and promoting the use of wood for energy. However, there is much debate still as to the carbon-offset efficacy of tree-planting in extra-tropical areas as compared to tropical zones.

Concluding remarks: key issues

This book addresses key issues in land restoration that emerge from restoration science and practice in the Mediterranean Basin. On the grounds of the long-standing and well-developed afforestation experience in Mediterranean countries, we shall suggest ways to incorporate lessons learned from past experiences into new restoration approaches, to face new and old threats, and new challenges and opportunities, using new and not-so-new approaches and techniques. The first part of the book deals with missing elements in the restoration practise that are critical steps for rationalising the incorporation of restoration activities in the economy, namely quality control, monitoring and evaluation. The second part tackles specific, innovative developments of restoration techniques. These include plant selection of species and provenances, and nursery and field techniques to overcome water stress as the major limitation in drylands – now and in the perspective of projected climate change. Specific chapters are devoted to developing restoration strategies and measures for widely representative cases of desertification-threatened lands, i.e., semi-arid lands subjected to long-term degradation under high water stress, and burned forests. In a final chapter, the editors offer a summary and synthesis, and some thoughts on the way forward.

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II.

The restoration process

Criteria for Recognizing, Organizing, and Planning Ecological Restoration

2

Andre Clewell, James Aronson, and James Blignaut

Introduction

In this chapter, we pose three questions. First, what criteria qualify a project as "ecological restoration" and distinguish it from other kinds of projects? Second, how can a project be organized, structured, and administered by its sponsoring institution or community in a manner that ensures that the distinguishing criteria for ecological restoration are satisfied? Third, what are the critical steps in planning a project so that it qualifies as ecological restoration and provides maximum value, including ecosystem services?

The concept of ecological restoration (sometimes called 'eco-restoration') has undergone substantial evolution during the past 30 years. It began as a simple proposition to put an impaired ecosystem back the way it was at a time when it was still whole, intact, and not degraded. At first glance, this seems reasonable enough, but it did not always work well, particularly at smaller spatial scales and at finer resolutions. The problem in "putting it back the way it was" assumed that nature was static and it could be repaired as if it were a work of art or architecture. Nature may appear static over the course of several years; however, natural ecosystems are dynamic, consisting of many living organisms belonging to a multitude of species that interact with each other, and respond to an ever-changing biophysical environment influenced by an ever-changing world and biosphere.

In addition, there was the problem of figuring out exactly what was meant by "the way it was" because historical records may be vague or absent. For example, the general trend of deforestation is obvious for a region like the Mediterranean Basin, but the historical ecological record is scant and generally inadequate for developing a model as the target of restoration at a specific site. To complicate this situation, people have been intimately involved for thousands of years in shaping our current ecosystems, to the point that several kinds of ecosystems can occupy one site, depending on past and present land usage (Blondel and Aronson 1999). This raises a problem because "the way it was" could be any of several different kinds of cultural ecosystems and some pre-cultural states. In addition, some landscapes have been modified by public works projects and other environmental alterations to the point that a former ecosystem could not possibly be put back "the way it was." Finally, even if it could be "put back," it may be liable to rapid modification from conflicting land use priorities among stakeholders.

When faced with these constraints, it is tempting to dismiss eco-restoration as quixotic and instead try to "put it back the way it formerly *functioned*". In other words, the species composition and community structure are less important than function. "Function" is shorthand for a suite of ecological functions, such as primary productivity, nutrient cycling, and food chain support. However, the principle meaning of "function" in this context, at least for most people, is the suite of ecological goods and services provided by intact, ecologically healthy ecosystems and that are of direct benefit for people. Examples of ecosystem goods are foods, timber, fiber, forage, thatch, fuelwood, and medicinals. Examples of ecosystem services are the provision of clean air and water; retention of flood waters; control of erosion; renewal of topsoil; enhancement of habitat for wildlife and rare species; sequestration of carbon, pollutants, and excess nutrients; pollination of crops; biological control of crop pests; and the fulfillment of human cultural needs of a spiritual, aesthetic, intellectual and recreation nature. People value ecosystems because they provide these goods and services.

The repair of ecosystem function is generally called *rehabilitation* rather than restoration. Rehabilitation poses its own, inherent disadvantages. One is that most rehabilitation programs are intended to resolve a particular problem, such as the recovery of grazing land or the provision of wildlife habitat. Site preparation and planting that are designed for one purpose generally provide only one appreciable service. In addition, ecosystems that are rehabilitated for a single purpose are readily susceptible to biological succession and may revert to their former conditions or change into states that were unanticipated and unintended.

The concept of ecological restoration has evolved considerably to the point that it fully embraces the restoration of function in the sense of rehabilitation, with the recognition that we can only restore to a *future* state. That future state may closely resemble the prior state, assuming that there have been no substantial changes in environmental conditions. However, global, regional, and local conditions are undergoing rapid change in climate and from sea level rise, as well as from multiple direct impacts of modern human activity. Consequently, the future state commonly develops under a new set of irreversible environmental conditions which temper the trajectory of an ecosystem to an altered state in terms of its species composition and community structure.

For example, irrigation may lower the water table over a broad region and cause irreversible changes in a natural ecosystem that has not otherwise suffered any abnormal stress or damage. If that same ecosystem were degraded, damaged, or destroyed from another impact such as overcast surface mining, and eco-restoration were performed for its recovery, then the target of restoration would be the expected future state with a lowered water table and not a former state that emulated the nostalgic past. In addition, restoration efforts would attempt to facilitate all ecosystem services of value to stakeholders and the restoration target could undergo further adjustment to accommodate the fulfillment of those values. Cultural modification of ecosystems for such purposes is not new and has been practiced globally for many millennia. In this new conception, 'restoration' becomes a powerful metaphor rather than an attainable reality, because only the future can be 'restored.'

We call this approach that addresses ecosystem states, ecological functions and service, and the satisfaction of human values "holistic ecological restoration" (Clewell and Aronson 2007). We recommend that all holistic restoration projects be conceived to help satisfy three "Rio Conventions" pertaining to the amelioration of climate change, reversal of desertification, and protection of biodiversity, which were adopted by the United Nations Conference on Environment and Development at the 1992 Río Summit (Blignaut et al. 2008). Eco-restoration contributes to climate amelioration by increasing carbon sequestration and providing vegetative cover that reduces the dissipation of solar radiation into the atmosphere as heat (Clewell and Aronson 2006). It also contributes to the reversal of desertification by recovering biotic community structure, and it returns biodiversity that had been lost. Eco-restoration simultaneously and synergistically addresses all three accords in some degree, depending on the availability of local institutions with appropriate environmental policies in place, sufficient scientific knowledge of the ecosystems being restored, and the local capacity to implement strategies and techniques for effective restoration. The degree to which a restoration project can address these Río Summit Conventions depends on local socio-economic, political, scientific, and technological conditions. Nonetheless, restoration should be conceived with all three in mind. It should also be clear that in our rapidly changing world, on-going maintenance or management will be required, in most or all cases, following completion or closure of a restoration site or project. In the next section we will explore these ideas in more detail, and in the context of specific projects.

Project Criteria

The question arises: What are the criteria that distinguish ecological restoration from another kind of project, for example, from afforestation? Actually, afforestation can serve as an important component in a restoration project, but not necessarily from the perspective of silviculture that is conducted exclusively for the production of wood products. In silviculture, competing native grasses and shrubs are commonly removed mechanically or treated with herbicide to reduce competition that could retard planted tree establishment. Only good quality nursery stock is planted to optimize favorable wood development. Measures are taken to protect the project site from potentially damaging fires. In a restoration project, native grasses and shrubs are protected, and more species may be introduced, even at the risk of reducing tree establishment and growth. A certain percentage of silviculturally inferior trees are acceptable, and indeed even desirable, for planting, because these may become ill-formed and provide denning cavities for animals. Fires, other than lethal crown fires, may be encouraged or even ignited to mimic historical conditions or to create spatial heterogeneity within the project site. In short, restoration projects may borrow heavily from other fields such as silviculture in terms of their methods; however, the intent of restoration has substantially greater amplitude than other kinds of projects. This example on afforestation serves to emphasize the importance of being clear on exactly what we mean by ecological restoration.

Ecological restoration is the process of assisting the recovery of an ecosystem that has been degraded, damaged, or destroyed (SER 2002). Ecosystems that have undergone such impairment have likely suffered from the ecological simplification of their community structure, or characteristic species were lost and replaced by more weedy, generalist species or by invasive non-native species. The impaired ecosystem may have lost some of its beneficial soil properties and the ability to recycle mineral nutrients efficiently, or the capacity to maintain and regulate a favorable moisture regime or microclimate. Restoration is the process of returning an ecosystem to wholeness in these respects and to a state of ecosystem health, much as a physician would heal a patient.

Once restored, the formerly impaired ecosystem should display these ecological attributes:

- appropriate species composition that is sufficient to allow development of normal community structure;
- absence of invasive, nonnative species, to the degree considered necessary to protect the health and integrity of the system;
- presence of all functional species groups or their likely spontaneous appearance later as the restored ecosystem matures;
- suitability of the physical environment to support the biota;
- normal ecosystem function or at least the absence of signs of dysfunction;
- integration with the surrounding landscape in terms of normal flows and exchanges of organisms, materials, and sources of energy;
- absence of external threats from the immediate landscape to the integrity and health of the restored ecosystem to the greatest practicable extent.

The restored ecosystem is resistant or resilient to frequent or common stresses and disturbances. Examples are winter freezes, summer droughts, periodic grass fires, and in coastal regions, exposure to saline aerosols. In addition, the restored ecosystem is self-organizing and therefore self-sustainable without evidence of arrested development. However, extreme events may disrupt even the most carefully conceived and executed project. In the coming century, this may be increasingly likely in the Mediterranean region, and elsewhere, as a result of anthropogenic climate changes (IPCC 2006).

The ecosystem should be restored with a target or model in mind, which is called a *reference ecosystem, reference model,* or *reference*. The reference reflects the desired state that the restored ecosystem is expected to approximate after it has attained ecological maturity. The reference can be one or more actual ecosystems, or it can be a representation of them, such as a published ecological description. It can be the historic ecosystem as recorded in photographs or museum specimens. It can be a remnant of the historic ecosystem that still persists on the project site or on a similar site nearby. It can be synthesized from a number of sources which collectively portray a reasonable approximation of historic conditions (Egan and Howell 2001). The reference model should accommodate recent environmental changes on site or in the vicinity that will influence future ecosystem states and the overall trajectory of ecosystem development in the spirit of holistic restoration and "restoring to the future".

Selection of the reference is critically important to restoration. Without a reference, the project lacks a valid starting point that reflects local environmental conditions and biota. A trajectory is equivalent to the wake of a boat which reveals where you have been and the direction you are headed. Likewise, the trajectory reveals where the ecosystem has been in the recent past in terms of its composition and structure. It also tells you the direction it is going, relative to its probable future state, from clues in the biota and in the current environment. In other words, it is the basis for the prediction of future conditions and thus the development of a realistic reference model for restoration. Without a reference, the predictive element in restoration is lacking, and it would be more accurate to call the project re-vegetation or ecological engineering rather than restoration.

Most terrestrial ecosystems are culturally modified or crafted in some degree. This is not surprising, because we humans comprise a dominant biological species, and we evolved in concert with the rest of nature. Cultural ecosystems, which include nearly all ecosystems that occur in the Mediterranean region, were 'sculpted' and redesigned by people, and most are still managed by them. Humans have domesticated or "gardenified" these ecosystems in the course of their transformation and the practical, intentional management of them, and of the larger landscapes in which they occur.

Of course, not all traditional management of ecosystems was sustainable in the past, and many of them caused land degradation. For example, humans who lived in small villages throughout the Alps modified the composition and structure of their forests as they harvested wood products and wild foodstuffs and introduced livestock and forage species. Such activities transformed alpine forests into cultural ecosystems that basically retained their productivity and forested aspect. However, land use intensified to the point that these forests were degraded or entirely destroyed, leaving bare and severely eroded slopes (Hall 2005). The selection of a reference for restoring an intact cultural ecosystem assumes that traditional cultural activities and land usage will be practiced, or that appropriate ecosystem management will be substituted for traditional practices, after restoration activities have been completed. Otherwise, the restored system may transform into another, less desirable state. It is critical to apply landscape perspective, as that is the scale at which most people perceive and communicate when it comes to issues of general, social interest. Sometimes, of course, the operative scale of an individual property – a farm or domaine – can be appropriate as well.

Organizational Criteria

Ecological restoration differs from civil engineering, architecture, landscape design, gardening, agronomy, silviculture, and related disciplines, on account of its being dynamic, open-ended, and lacking a static end-product. In these other disciplines, the product is carefully molded to specifications that are clearly prescribed in plans and drawings, whether it is a bridge or building, grain field or row-planted forest, etc. The product of ecological restoration is nature. Nature by definition is distinct from human artifice and cannot be authentic if it is molded to fit a preconceived notion. The process of ecological restoration merely initiates or accelerates natural processes in a manner that allows nature to recover and to heal itself.

Sponsors of ecological restoration projects are commonly governments or transnational agencies like the European Union, World Bank, or United Nations. Some projects are sponsored by non-government organizations (NGOs) such as the World Wildlife Fund or Conservation International. In some parts of the world, restoration is sponsored by local community-based organizations (CBOs) with local government, business, or philanthropic support. Yet others are sponsored by tribal communities on commonly held lands in remote rural regions with the assistance of agency personnel or other experts. A few projects are conducted by individual land owners and managers on their own initiatives.

Governments, transnational institutions, and larger NGOs commonly sponsor a variety of kinds of activities, probably all of them (except for restoration) with a clearly described end-product. The production of that end-product is ensured if the project sponsor generates carefully conceived plans and maintains careful quality control throughout the life of the project. Activities are conducted by technicians who must adhere closely to design criteria, contract specifications, and sometimes regulatory standards. The final product can be weighed, tested, or subjected to other empirical measures and evaluated according to criteria specified in project plans.

This approach is inappropriate for ecological restoration projects, where the end-product consists of a dynamic ecosystem consisting of living, interacting organisms and cannot be predicted with precision, at least at smaller spatial scales. The rationale for restoration is not to produce a single – or very few – services or products, such as a specific crop. Instead, ecosystems are restored to fulfill a wide array of tangible and intangible products and services. Nonetheless, larger institutions are managed in accordance with their internal protocols by professionals who may have never visited a restoration project site. Consequently, restoration is commonly treated as if it were an engineering function using steel, concrete, and other inert materials.

The product of restoration –an intact functional ecosystem– is a long-term investment of land and effort that must fulfill the values of those who stand to benefit from it. These values are personal and cultural, objective and subjective. They include ecological services of economic consequence, and they satisfy cultural needs such as serving as outdoor venues where ecological literacy of school children can be elevated. They are reservoirs of biodiversity. They allow restoration practitioners to find satisfaction in repairing what a previous generation had destroyed. The fulfillment of these and other values may be as important as the restored ecosystem itself, and some values could continue to provide fulfillment indefinitely for generations. Many values are not amenable to empirical measurement, and others can only be estimated indirectly. Therefore, the importance of all restoration projects that are conducted according to an engineering paradigm is necessarily underappreciated.

Obviously, sponsoring organizations must apply very different approaches towards conducting restoration and appreciating its results. One way would be to determine to what degree each of the seven bulleted items listed earlier in this section have been attained. Another would be to compare a completed restoration site with its pre-project condition or, alternatively, to its reference model, making allowances for differences in ecological age between them. A third approach is to develop short-term objectives which, if reached, would signal early development that should eventually lead to the achievement of project goals as indicated by the reference model. Another approach is to apply sociological criteria to evaluate the attainment of values. The approach to evaluation of a restoration project must be nuanced and requires sophistication that reflects project objectives and goals. If, for example, restoration was conducted to compensate for specified environmental harm, then the approach to evaluation should be to demonstrate compliance with relevant norms. If the rationale for restoration was to satisfy more broadly conceived cultural values, then the criteria should measure progress towards longer-term goals (Zedler 2007). An exercise of evaluation for forest restoration projects is developed in Chapter 4.

The conduct of the project also requires a different approach. Preconceived design criteria may prove to be ineffective, and mid-course corrections may be needed. Project goals may not be served by a strict technological approach. Instead, the restoration practitioner needs to have leeway for what Aldo Leopold (1949) called “intelligent tinkering”. Ecosystems are quite complex, and they develop sequentially in a milieu of variable environmental conditions. A perceptive practitioner can “read” the landscape and administer corrections for problems that could not have been anticipated in the project planning stage. For this reason, project management requires flexibility, and project budgets should include funds for contingencies that can be made available quickly.

Planning Criteria

Restoration projects may seem simple enough, but they can be disarmingly complex. Scheduling, for example, requires knowing when weather and other conditions are favorable for each step in site preparation, when seeds can be gathered, and how long it will take for nursery stock to reach its prime for out-planting, as well as the usual complications to secure equipment when it is needed and to muster labor. The planning process includes an inventory of the project site prior to restoration in order to document its condition, determine appropriate strategies and methods of restoration, and later to assess the efficacy of restoration. Stakeholders should be notified and engaged in the project to the greatest possible extent. Otherwise, the value of the project may not be appreciated by local citizens, and it will garner disrespect instead of protection and stewardship. Pre-project monitoring of hydrological or other environmental conditions may be necessary in order to establish a baseline for project planning and evaluation. Selection of the reference model is another task that must be completed before project goals can be finalized and planning begun. In short, there are many steps to a restoration project. Ignoring any of them can be costly both in terms of time to completion and funding.

Recordkeeping is of considerable concern. A thorough photographic record is essential, including many photos of the project site prior to the implementation of project activities. Otherwise, the achievement and significance of restoration may be lost to all but a few practitioners. Other recordkeeping is helpful for orienting new personnel who are engaged midway through a project that could take a decade or more to complete.

Government agencies, transnational institutions, and larger NGOs commonly engage project planners who are not necessarily restorationists and who may never have the opportunity to work at the project site. Instead, they develop layers of GIS maps and use standard landscape designs and software to prepare as-built images. If a design group is engaged, they should work closely with the restoration practitioners who will be engaged to fulfill project plans. This is not always done, and the practitioners may not even be hired until the plans are complete. Practitioners can anticipate problems and conditions that may be invisible to professional planners. Their collaboration can prevent costly problems that could arise during project implementation.

The Society for Ecological Restoration International (SER) has developed a checklist and summarized scheme of the steps for any ecological restoration project (Clewell et al. 2005). This document is available on the SER website (www.ser.org), and was printed verbatim as an appendix by Clewell and Aronson (2007). We recommend that this checklist be employed in any ecological restoration project to ensure that all steps are undertaken and that none are missed. The document is designed for use by personnel at every level, from directors within the sponsoring organization to project managers and restoration practitioners in the field. Specific restoration techniques that are applicable in the Mediterranean region are described by Whisenant (1999) and Bainbridge (2007) (see also Chapter 8, this volume).

Some projects are local initiatives by CBOs, tribal councils, or individual property owners or managers. Such projects have the advantage that restoration practitioners have responsibility, authority, and control over all aspects of a project, and they are not beholden to an administrative hierarchy that can hinder project implementation if unanticipated events or misunderstandings occur. Furthermore, practitioners can work collegially to develop contingency plans or conduct “intelligent tinkering”. In addition, community based projects are particularly amenable to stakeholder participation and the development of stewardship organizations that will provide the completed project with protection, local management, local use and appreciation, and local political support. However, locally sponsored projects are commonly under-funded and hindered by inaccessibility to equipment and expertise and are necessarily small-scale and not particularly complex.

A better model, which already has some precedence and is worthy of serious consideration, is what we call the “inside-out” approach. Instead of a technocratic “top-down” approach whereby governments and other large institutions impose a bureaucracy that reduces the role of the practitioner to that of a field technician, or a “bottom-up” approach whereby CBOs are strapped by insufficient funds, expertise, and equipment, the “inside-out” approach combines the benefits both without retaining their drawbacks. The "inside-out" approach is so-designated because local people are working from within their own ecosystems to restore them (Waltner-Toews et al. 2003). Projects are established locally by CBOs as local initiatives and conducted primarily by local restoration practitioners. Large institutions (government agencies, transnationals, larger NGOs) enter into partnership with the CBOs to lend financial support, expertise, and specialized equipment, and also to provide a regional perspective to which the local project contributes. In this manner, these larger organizations or institutions operate in more of a collegial manner. The CBO is allowed some leeway to make its own mistakes and take corrective measures. The larger entities in the partnership can step in and rescue a project, at least temporarily, if the CBO or its restoration team falters or collapses for any reason. Stakeholder interest and engagement is optimal under this arrangement, and the values generated by a restoration project are maximized.

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Economic, Social and Cultural Factors Affecting Landscape Restoration

3

David Lamb

Introduction

Many ecologists see the primary objective of restoration as being the re-establishment of biological diversity on degraded lands. It seems self evident that this is a worthy goal. By doing so biological diversity is conserved and the key ecological processes that are necessary for the functioning of ecosystems are, hopefully, restored. But ecological restoration is often difficult because the nature of the original ecosystems may be unknown or impossible to achieve because of historical events. For example, certain original species may have become extinct while new, non-native species may have become naturalized and invasive. It may also be difficult to define or reproduce the precise assembly rules needed for restoration to proceed. Additionally, of course, the cost of restoration can be found to be prohibitive, especially if the cost is borne unequally among the stakeholders.

However, there is a more fundamental impediment to restoration which, if not addressed, will limit the extent to which it can be implemented over large areas. That impediment is the potential it can have to adversely affect certain landholders in the region being restored. This may seem paradoxical. After all, the purpose of restoration is to improve and not hinder human livelihoods. The difficulty lies in what can be thought of as the distributional effects of restoration – the community as a whole may benefit but the direct and indirect costs of carrying out restoration may be paid by a much smaller number of individuals such as those owning or using the land that is treated. Under such circumstances it may be very difficult to persuade these land managers to participate in the restoration of large areas of degraded land even when the technical means to do so are available.

This chapter has two objectives. The first is to examine how such socio-economic circumstances as well as biophysical conditions can affect restoration options. The second is to describe some methods that might be used to evaluate the socio-economic consequences of restoration such that managers might revise their approaches and adapt to unexpected circumstances if this becomes necessary.

The landscape mosaic

Landscapes are not uniform. Rather, they are made up of a mosaic of vegetation types and land uses (Gilmour 2005). For example, many landscapes contain areas of relatively undisturbed "natural" vegetation as well as intensively cultivated agricultural land. There may also be areas of more disturbed native vegetation and areas of less intensively used agricultural land such as grazing land (Table 1). These differences reflect the changes induced by past events and management decisions as well as those caused by current disturbances such as grazing or fire.

Different parts of this ecological mosaic will deserve restoration treatments more than others. The most obvious areas for restoration are those that are highly degraded such as steep slopes, saline areas or eroding stream banks. Other sites that might deserve attention could be the often large areas of poor quality agricultural land surrounding patches of natural vegetation. There is unlikely to be any debate about the value of overcoming erosion or salinity but there may be some concerns about intervening to change current land use activities on agricultural land even when this is of poor quality. This is because such land may still be in use. Any changes in its use will have an opportunity cost.

TABLE 1. The ecological mosaic and its influence on restoration choices.

Bio-physical unit	Influence on restoration choices
Intact native vegetation	These areas are likely to be of greatest ecological significance because they are reservoirs of biodiversity and a source of species to recolonise degraded landscape areas.
Disturbed native vegetation	May still contain significant amounts of biodiversity but may also contain weeds and pest species. Often regularly burned by wildfires. Potentially available for restoration because any opportunity costs are likely to be small.
Intensively used agricultural land	This land is unlikely to be available for restoration because the opportunity cost would be too large.
Less intensively used or degraded agricultural land	This land may appear to be available for restoration and have a low opportunity cost. In fact it may be being (unofficially) used by some individuals or community groups and these would bear the cost of restoration.
Riverine areas	Important for biodiversity conservation but often degraded by graziers and other land users. Erosion from such areas may impact heavily on downstream stakeholders.
Unstable hillslopes, saline areas or otherwise highly degraded sites	High priority for restoration. Unlikely to be used by landowners and may be adversely affecting other downstream stakeholders
Fire-prone areas	The fire regime may vary across the landscape with some areas being more frequently burned than others, and therefore deserving different prevention and restoration efforts.

The type of restoration undertaken is also likely to vary with the type of land being treated. In the case of disturbed but mostly intact native vegetation, it may be possible to foster biodiversity by simply protecting the site from further perturbations. The costs of doing this may be relatively modest. However, this approach might not be possible on more degraded sites. In these cases the cost of restoration is likely to be higher and the more feasible objective may be to restore ecological processes and functioning rather than biodiversity. It is also likely that the treatment might be required to generate a direct financial benefit to justify the expense.

These land use patterns are, in part, a consequence of the socio-economic mosaic that overlays the ecological mosaic. Thus land might be owned by public and private landholders. It will also be distributed among landowners who differ in the sizes of the properties they own, in wealth and in political influence. Some farmers may have secure tenure for their land while others may not although they may believe they do so. Some farmers may not own land they use but may have long-term rights to use it for grazing or other purposes. In some cases there may be disputes over ownership of a particular area of land. Property rights are likely to be more strongly asserted over productive agricultural land than over degraded lands. On the other hand, some degraded lands may be treated as common property resources that are available to anyone for purposes such as grazing. In such cases further degradation is inevitable. The ability to intervene and undertake restoration in the various land units described in Table 1 will depend very much on who owns or uses the land and how these people fit into the various landowner or land user classes referred to above.

But, in addition to these on-site land managers there may also be other stakeholders with legitimate interests in the way these lands are managed. These might be downstream water users such as hydro-electric authorities or people living outside the region but interested in wildlife conservation. They may also include adjoining landowners affected by weeds, fire or erosion coming from the area.

This socio-economic mosaic and extended list of stakeholders has several consequences. Firstly, there are likely to be significant differences in the capacity of landowners and managers to undertake restoration or even prevent further degradation. Poorer farmers are more likely to be affected by fluctuations in agricultural prices or by periods of drought than are wealthy farmers and this will limit their capacity to act. Secondly, there are likely to be quite diverse views amongst the various stakeholders about where and how restoration should be carried out. Thus a farmer facing uncertain prices for agricultural produce is likely to be less concerned about protecting biodiversity than a city-based wildlife enthusiast with a secure job who is immune to market fluctuations.

Incorporating restoration into existing land use patterns

As mentioned above, many landowners may be reluctant to change their current land use practices even though these are causing land degradation simply because they cannot afford to do so. Others may be reluctant to act because they fail to recognize that degradation is occurring (or because the adverse consequences are largely occurring off their property). A third group may be reluctant to change because the proposed new land uses are seen as being too complex or radically different to those currently being practiced. This may be the case with many restoration systems that involve new biological communities, rather different management systems and which generate a quite different set of benefits to those usually experienced by most farmers. The fundamental task, therefore, is to develop ways of making these new systems sufficiently attractive to farmers and other landholders such that they will adopt and maintain them.

Some of the factors that may influence the attractiveness of restoration to landowners are shown in Table 2. Several of the most important factors involve land. Perhaps the most important of these is land tenure. Farmers without secure rights are unlikely to undertake a long-term venture like restoration because they have no way of ensuring that they will benefit from doing so. For those with tenure the area of their farm becomes important. Farmers with large amounts of land or with large areas of unproductive or relatively inaccessible land are more likely to be willing to undertake restoration on at least part of this land than those with only small landholdings or land that is, in their eyes, fully productive. The issue here is the opportunity cost of restoration. What are the economic opportunities forgone by taking land out of its existing use for restoration purposes (e.g., to create a corridor between two forest remnants)?

Sometimes landowners are legally obliged to undertake some form of restoration for the public good (e.g., to eradicate noxious weeds, maintain a particular fire regime, to revegetate and stabilize stream banks). In other case they are assisted in carrying out restoration for the public good by financial incentives or subsidies to carry out restoration. Certain forms of restoration may generate direct commercial benefits such as when there are payments for the goods (e.g., timber produced by new plantations) or the services (e.g., clean water, wildlife habitats) produced by restoration. Payments that are received in the short term are likely to be valued more than those that only arrive after some years.

The likelihood of a landowner receiving payments from goods or services generated by restoration may depend on just how degraded the landscape might be. For example, there may be rather less likelihood of a strong market for farm grown timber or non-timber forest products if there are large natural forest areas nearby that are already able to supply these. This situation may be reversed in a highly degraded landscape with little of the original forest remaining. In such a case the market prices for goods or services may be much greater. This means that a restoration system that can generate these goods and services is more likely to be attractive than one that does not.

In many parts of the world farmers derive a significant part of their income from activities off-farm. They may work for other farmers or may be employed in nearby towns. This means there is less need to achieve a commercial outcome from agricultural activities. Consequently, there may be greater scope for restoring at least part of the landholdings.

TABLE 2. Socio-economic factors that may influence the attractiveness of restoration to farmers and landowners.

Factor	Significance
Land tenure	Farmers without secure land tenure or usufruct rights are unlikely to undertake a land use activity such as restoration where the benefits take time to achieve.
Availability of agricultural land	The commercial viability of a farm will often depend on its size. It may be easier to undertake ecological restoration on a large farm than on a small farm because the initial impact of the change is proportionally smaller.
Productivity of land	Restoration will be more attractive on land that is regarded as unproductive (because of lost soil fertility, weeds, pests etc.) than on land that is still highly productive. This is because the opportunity costs incurred in converting productive land would be too high.
The likelihood of financial or other direct benefits arising from restoration	Landowners are more likely to be interested in a land use activity that benefits them immediately and directly. Benefits may come from goods such as timber or services such as improved water supplies or new wildlife habitats.
Availability of subsidies, incentive payments or tax concessions	Such payments may be especially significant for small, risk-averse landowners or those with low incomes.
Legal obligations to overcome degradation	There may be legal requirements on landowners to prevent fires or eradicate weeds or pests.
Availability of alternative sources of off-farm income	Landowners able to obtain income from off-farm employment may be more able to convert part of their land holdings to new uses such as restoration.
Attitude of neighbours	Neighbours can have positive and negative influences. Innovative neighbours can provide examples to be copied but conservative neighbours can also argue against change and diminish the propensity of innovators to take on risky new land uses.

Pannell (1999) has identified four key conditions that are necessary for a farmer to adopt a new land use. The first of these is that a farmer must be aware of the innovation. Being told about it is rarely sufficient. A rather more powerful introduction is to actually see the innovation in use. Field demonstrations of restoration that clearly show the benefits of the new system can be very useful (especially if these benefits develop quickly and have a cash value) and demonstrations on the land of a neighbour with similar field conditions may be especially persuasive. Secondly, there must be a perception by farmers that it is feasible to test the innovation on their land. A complex change involving the planting of large numbers of seedlings of different species may be rather less attractive than, say, a change simply involving fencing to limit grazing pressures or a change in fire regimes. Thirdly, the innovation must be feasible but it must also be seen as being of low risk and sufficiently promising to be worth testing in a small scale trial. Finally, even at this early stage, it should be clear that the innovation will promote the farmer or landowner's overall objectives. That is, it should be clear that it will be in the self interest of the landowner to make the change. Though the relative financial profitability of alternative land uses drives many farming decisions, farmers are often motivated by other factors as well. For example, many landholders may be keen to overcome environmental degradation for their own and for the community good. Others may be motivated by a sense of stewardship or a desire to be seen as good land managers.

Rather different and sometimes more complex arrangements might be needed where restoration is undertaken as a community activity rather than by individuals. This might occur when a community acquires the rights to administer a village commons or people group together to manage a local watershed area. In such circumstances the community may have acquired control of land previously being degraded by unregulated use (e.g., the disturbed native vegetation or less intensively used agricultural land of Table 1). Forests are often able to regenerate once such areas are no longer burned, grazed or logged meaning a new economically valuable resource is created. A common approach is for the community to take advantage of the new forests and establish management rules that regulate who can access the land and the extent to which any resources (e.g., timber, pasture) can be used (Gilmour 1990). Unlike the situation described above involving individual farmers, the opportunity costs in this case are mostly very low.

Case studies

Case Study 1: Appropriate policy settings can facilitate the restoration of degraded landscapes

The natural vegetation of the Shinyanga region of north west Tanzania is mainly miombo woodland and acacia scrub and has a rainfall that varies between 650 – 1100 mm. The lands are used by pastoralists who practice a form of communal grazing. These people traditionally maintained a series of enclosures (5-100 ha) to provide fodder during dry periods as well as supply other products such as thatch material, medicines and firewood.

Degradation occurred throughout the region following a period of deforestation in the 1920s and 1930s which was aimed at eradicating tetse fly. Large areas of woodland were also converted into agricultural land for cash crops such as cotton and rice. The traditional grazing practices were further disrupted after 1975 when many people were relocated as part of the Government's "villagisation" program. This sought to improve the provision of government services such as education and health by forcing people into special villages where they could be more easily contacted. But this change destroyed local management practices. By 1985 the traditional land management systems had virtually died out and the system of maintaining enclosures was abandoned. Overgrazing and degradation became widespread throughout the region. In 1986 President Nyerere called it "the desert of Tanzania".

After the mid 1980s the villagisation scheme was abandoned and the government adopted a new policy of decentralization. Communities were given tenure over their lands and the old enclosure and management systems were revived. This allowed natural vegetation to recover even in what appeared to be highly degraded sites. Tree planting was also encouraged. The result has transformed the landscapes and fostered a remarkably widespread recovery of woody vegetation across an area of some 250,000 ha.

The main lesson is that large scale restoration can be possible at a very low cost if appropriate policy settings are established. In this particular case the government recognized that traditional community management practices had previously allowed a stable form of land use to evolve and that it needed to empower these traditional institutions once more to enable them to re-establish these practices (Source: Barrow and Melenge 2003, Wood and Yapi 2004).

Case Study 2: Degradation can only be overcome if there is a sharing of costs and benefits among the various stakeholders

Salinisation has occurred in parts of southern Australia following the clearing of some of the original forests and woodlands. The replacement of deep-rooted native species by shallow-rooted agricultural crops changed the hydrological cycle by causing a reduction in evapo-transpiration. This has allowed saline ground waters to rise close to, or even reach, the soil surface. Large areas of previously productive land have been adversely affected. In some landscapes it is possible to reverse these changes by planting fast-growing tree species in recharge areas (e.g., on hillslopes). These trees increase water useage and cause water tables to decline thereby reducing salinisation in discharge areas (e.g., in valleys). Trials have been carried out to explore just which landscape units should be planted to generate the greatest benefit. These generally show that the more a watershed is reforested the greater the hydrological improvement (Schofield 1992).

But the technical success of tree planting is not necessarily sufficient for it to be widely adopted. Trees can help overcome salinisation but they are costly to establish and they also replace existing land uses such as cropping. This has financial consequences. It may be possible to grow trees for some commercial benefit but, unlike annual agricultural crops, the financial returns are less frequent. Many farmers may be unable to afford to convert a significant area of their farm to trees and still remain financially viable. The dilemma increases when more than one landowner is involved. It is often the case that the land use practices causing salinisation are some distance away from where the effects become evident. This means that salinisation in one area belonging to Landowner A (e.g., in a valley) may be induced by land clearing and other activities on another area (e.g., upslope) belonging to Landowner B. This poses a particular dilemma. Why should Landowner B replant trees across his land for the sake of Landowner A? Further, if he is to be compensated for doing so, should it be only Landowner A who pays or should other downstream water users (e.g., townspeople) who are now receiving salty water also pay?

The key lesson is that there are usually a large number of people concerned with, and affected by, the management of degraded lands. These include those actually using the land as well as neighbours and others who may live some distance away but who have a legitimate interest in how the land is managed. If rehabilitation is to occur there must be a way of ensuring that the costs are shared amongst those who benefit from the change (Source: Walsh et al. 2003, Pannell and Ewing 2006, Environment Australia undated).

Case Study 3: Degradation can cause ecosystems to move to a new state condition. It can be very costly to reverse such changes

The rangelands of the states of Queensland and New South Wales in Australia have a rainfall of less than 600 mm. This rainfall is highly variable and drought is common. Prior to the arrival of graziers in the second part of the nineteenth century these areas were occupied by aboriginal people. The aboriginals were hunter-gatherers and frequently burned some of these lands as part of their hunting efforts as well as for other purposes. The fire regime changed and the frequency of fires greatly diminished when graziers arrived with their herds of sheep and cattle.

The new herbivores have reduced grass biomass. This decline, plus the absence of fire, has allowed many woody plants to regenerate. These woody species include native trees such as *Eucalyptus* or *Acaci*a as well as native shrubs such as *Eremophila* spp. and *Cassia* spp. These new woody species have, in turn, helped shade out more of the remaining grasses. The outcome has been a major change in the balance between grasses and woody plants and has led to a significant reduction in the supply of pasture to the sheep and cattle herds.

An intense fire would help control the woody plant populations but the new ecosystems do not normally have enough fuel to sustain such a fire. Sufficient fuel can be produced after an above-average rainy season but these good seasons are rare in these regions. Under such circumstances many graziers would rather use the pasture generated in these rare good periods to feed their stock rather than as a fuel to reduce woody plants.

The situation represents a significant management dilemma. From the graziers point of view the new systems are becoming degraded since they produce less pasture. If woody plants continue to encroach then many farms may become unprofitable since they will have too little pasture. The obvious solution is to re-introduce fire to exclude woody plants and favour grass. But some graziers may already be in the situation where they cannot afford to burn the additional pasture (i.e., fuel) provided by an above-average rainy season but must use it to feed their stock in order to pay their accumulated debts. But if they do not burn the situation will only get worse.

The key lesson is that economic circumstances can prevent degradation from being overcome even when the ecological knowledge necessary to restore a site is available (Source: Daly and Hodgkinson 1996, Burrows et al. 1990, Burrows 2002).

Case Study 4: Not all technical solutions are necessarily appropriate

Many of the savannah lands in the Lake Chad Basin of northern Cameroon have become degraded because of firewood harvesting, overgrazing and cotton farming. The result has been widespread degradation.

Trials have been carried out to test various ways of combating this degradation. These trials have generally sought to limit water run-off and conserve soil and have included a variety of earthworks including ploughing and various kinds of small dams and barriers to water flow. Tree planting using both native and exotic species was carried out within these treatment areas.

Most of these trials have been technically successful and the trees have flourished although there were differences between exotic and indigenous species in terms of survival and growth rates. However, it seems unlikely these technical solutions will be widely adopted. Most of them involve heavy machinery such as tractors or bulldozers and are simply too expensive for local farmers to adopt. Further, the tree planting methods used were able to incorporate a variety of commercially useful products such as fuelwood, fodder and medicines etc. but they did not allow for the incorporation of food crops.

The main lesson is that technical solutions alone are not sufficient to overcome degradation. Ways must also be found of incorporating these solutions into existing farming practices such that land managers can afford to adopt them (Source: Wood and Yapi 2004).

Monitoring change and measuring success

It is a relatively straight forward matter to monitor the bio-physical changes caused by restoration. This can be done by measuring attributes such as changes in plant cover, tree growth rates, plant species composition, hydrology or movements in wildlife as new habitats develop. It is a rather more difficult task to monitor change and measure the socio-economic "success" of restoration. As noted earlier, there may be a large number of stakeholders who may react quite differently to a restoration or rehabilitation project with some judging it a "success" and others counting it a "failure". The problem is in deciding how to take account of these different views and reactions. This problem means it is difficult to be prescriptive about the ways monitoring should be carried out. Different methods will be needed in different situations. None the less, certain indicators may be more generally useful. These are outlined in Table 3.

One obvious indicator is the economic circumstances of people living in the area. Restoration is unlikely to be possible if household incomes are declining. In fact these circumstances are more likely to promote further degradation. This was the situation in Case Study 3 involving undesired woody plants. Rather "success" is more likely if incomes are increasing. This improvement may come from increased agricultural productivity (because of soil conservation), from tourism or from the sale of goods such as timber or grazing rights. Success is likely to foster further success and enable larger areas of degraded land to be treated.

A further promising sign would be the willingness of external stakeholders to pay for ecological services such as clean water derived from restoration. In the situation described in Case Study 2 a payment by downstream water users might be sufficient to compensate a landowner for reforesting part of their land to restore the former hydrological cycle. In such a case it may take time before the benefits of restoration become evident but monitoring of groundwater levels would provide evidence that positive changes were underway.

A third indication of "success" would be evidence that local communities continue to protect and maintain restoration areas in the expectation that the process will eventually benefit them. This was the situation in Case Study 1 in Shinyanga, Tanzania where the extent of the recovery was not clear when the process started. On the other hand, there was considerable traditional knowledge, especially amongst older people, about how these systems functioned and this would have provided some confidence that restoration was possible.

Strong evidence of support would also be provided by spontaneous new restoration projects initiated at other sites as a consequence of the benefits generated by earlier restoration efforts. Similarly, the development of new business enterprises that engage in the

TABLE 3: Socio-economic indicators of the success of restoration activities

Indicator	Reason
Incomes of resident households improving.	Farmers with declining incomes are unlikely to be able to afford to implement or maintain restoration activities.
Payments being made for ecological services.	External stakeholders willing to pay land managers for on-site restoration activities that yield ecological services such as clean water, biodiversity or wildlife habitats.
Individual landowners continuing to protect and maintain restored sites Spontaneous restoration at other sites by individual landowners without the need for external subsidies or support	Landowners continue to view restoration on their land as being a valid and beneficial land use. The benefits of restoration are self evident to individual landowners resulting in increased areas of degraded land being treated. The more of such new sites the greater the "success". A corollary of this is that no new areas of degradation are evident.
Development of private enterprises able to carry out or benefit from restoration activities (e.g., seedling nurseries, reforestation companies, weed control companies, ecotourism groups etc.)	Restoration and the land uses it fosters have become commercially profitable and created employment and new economic opportunities.
Development of institutions and learning networks amongst landowners aimed at fostering rehabilitation.	These institutions and networks generate, accumulate and transfer knowledge about restoration and the ways it can be incorporated in local land use systems.
Validation and community support for policies, regulations and institutions designed to protect restored areas and prevent future degradation	These regulations may be formal government legal regulations, traditional community regulations or new rules established by communities to facilitate the management of newly acquired common property resources. The institutions may be traditional community organisations or government regulatory bodies.

restoration program are also likely to be indicators of “success”. These might be seedling nurseries, tree planting groups, weed control groups or businesses specializing in removing animal pests. As restoration matures there might also be opportunities for eco-tourism. There was no evidence of any landholders at the Lake Chad sites described in Case Study 4 spontaneously adopting the restoration techniques being tested. This was because the approaches were simply too expensive for any individual to adopt even though there was evidence that they could work.

A key indication that restoration is likely to be a long-term activity independent of external support is the development of local organizations and institutions able to generate, accumulate and transmit knowledge. These institutions may revive traditional ecological knowledge systems and incorporate this knowledge with modern scientific knowledge and that gained by actually carrying out restoration (e.g., Case Study 1 at Shinyanga). Restoration is, ideally, a process of adaptive management but there should be systems or organizations able to collate and synthesise knowledge and make it more widely available.

Finally “success” might be indicated by the continued community support for policies, regulations and institutions designed to protect restored areas and prevent future degradation. These devices are likely to be the primary means by which individuals are prepared to sacrifice short-term individual benefits for the sake of large scale collective action that benefits the community as a whole. Without credible instruments like these this trade-off might be impossible to establish.

Conclusions

Degradation has many causes but it is often the result of adverse social and economic circumstances affecting individual land users. These circumstances have to be changed if degradation is to be stopped and the site restored. This means that ways have to be found to enable land users to include the necessary changes within their current land use plans if restoration is to proceed. The ways in which this might be done will depend on the degree of degradation that has occurred, the nature of the landscape mosaic and on the socio-economic circumstances of the particular land managers.

All restoration necessarily involves some adaptive management since it is rarely possible to forecast just what will occur over time. Such management requires feedback to indicate if a successful trajectory is being maintained. There are a number of socio-economic indicators of “success” that might be used including that peoples livelihoods are improving and that restoration is being spontaneously taken up by new land managers without the need for further external support. Perhaps the key indicator, however, is that institutions and learning networks have evolved enabling different experiences to be integrated, synthesized and shared amongst practitioners (Berkes et al. 1998). Such learning networks will assist these communities to withstand future ecological or economic shocks and prevent further degradation.

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Evaluation of Forest Restoration Projects

4

Susana Bautista and José Antonio Alloza

Introduction: The need

There is a consensus on the need for the evaluation of restoration and management actions (see for example, Clewell and Rieger 1997, Holl and Cairns 2002, Machmer and Steeger 2002, Thayer et al. 2003, SER 2004, Vallauri et al. 2005). The lack of evaluation and subsequent dissemination of the results of restoration actions limits the application of the best technologies and approaches available. Restoration treatments and techniques are often applied without questioning their efficacy. The cost-effectiveness of the restoration actions, particularly in relation to varying environmental and socio-economic conditions, remains poorly documented. The practice of restoration requires much better use of the existing restoration expertise and information, as well as improved understanding on the impacts of restoration strategies on the target socio-ecological systems.

Evaluation is the key element linking restoration practice and the advances in restoration science and technology (Fig. 1). The practice of restoration provides useful settings for tests of ecological and restoration theory (Bradshaw 1987, Jordan et al. 1987, Young et al. 2005). Similarly, the evaluation of the cost-effectiveness of new techniques across a number of real-world restoration projects provides the framework for technological advance. Moreover, monitoring and evaluation are critical components of an adaptive management approach to restoration (Murray and Marmorek 2003, Vallauri et al. 2005, Aronson and Vallejo 2006). For a given restoration action, evaluation provides feedback for the fine-tuning of the treatments and techniques applied, and thereby helps address the uncertainty inherent to ecosystem dynamics (see Chapter 5, this volume). For the general practice of restoration, evaluation helps managers learn from past restoration efforts and adapt restoration strategies and techniques in response to spatial and temporal variation in environmental and socio-economic conditions. Evaluation is needed to establish cost-effective thresholds for the various management alternatives, and to identify priority areas where actions could be most effective. Last but not the least is the two-way connection between restoration practice and society through evaluation, which provides both the

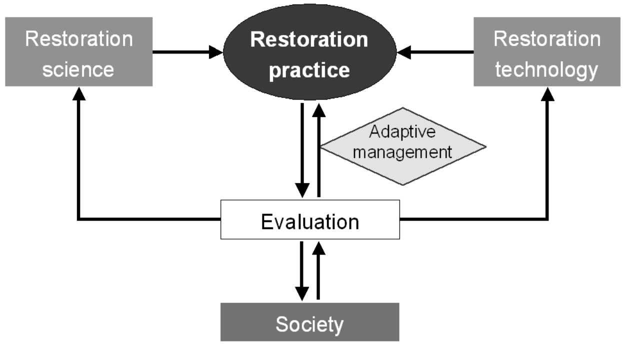

FIGURE 1. Schematic view of the linkages between restoration practice and restoration evaluation.

currency for disseminating the results and benefits of restoration and a way for incorporating social demands and perspectives into the restoration process.

Despite the unquestionable benefits associated with the evaluation of restoration actions, the actual number of restoration projects that are evaluated remains very low. Brooks and Lake (2007) examined records for 2,247 stream restoration projects in Australia and found that only 14% indicated that some form of monitoring was carried out. Berndhartd et al. (2005) reported than only 10% of >37,000 river restoration projects across the United States document any form of project monitoring, and little of this information is readily available for assessing the ecological effectiveness of restoration activities. Similarly, reforestation projects in the northern Mediterranean are rarely monitored and assessed (Bautista et al. 2010). As a result, restoration expertise remains under-utilised, hindering our capacity to incorporate what has been learned into future decision making. A number of recent review studies have addressed the need to evaluate the effectiveness of restoration actions (e.g., Maestre and Cortina 2004, Gómez-Aparicio et al. 2009, Rey Benayas et al. 2009, Bautista et al. 2010). These studies have provided useful information on the impacts of restoration on biodiversity and ecosystem functioning and have helped identify biotic and abiotic factors that determine the ecosystem response to restoration. However, as valuable as these independent studies can be, only regular feedback from the systematic evaluation of restoration projects provides the necessary inputs for adapting restoration strategies and techniques in response to environmental and socio-economic changes. Project evaluation should therefore be an integral component of any restoration action, and it should incorporate the active participation of managers and other restoration actors in the evaluation process.

Factors that impede incorporating evaluation into restoration efforts include the lack of long-term management programs for the restored areas and the all too common acritical assumption of theoretical paradigms (Cortina et al. 2006). Moreover, a more widespread and effective evaluation of restoration actions requires more work in developing, testing, and harmonizing evaluation tools and criteria (Aronson and Vallejo 2006). This chapter addresses this challenge by reviewing and discussing the state of the art on restoration evaluation, and presenting an integrated assessment protocol tailored to the long-term evaluation of forest restoration in the Mediterranean basin. Although most of the approaches discussed here are applicable to any type of restoration project, the chapter focuses on the evaluation of forest and dryland restoration to combat desertification.

Evaluation approaches

The approaches for evaluating restoration actions are many, including, among others, comparisons between restored and non-restored areas or between restored and reference target areas (Brinson and Rheinhardt 1996, Gaboury and Wong 1999, Rey Benayas et al. 2009); comparisons with natural range of variability (Hobbs and Norton 1996, Parker and Pickett 1997, Allen et al. 2002); degree of achievement of restoration goals (Zedler 1995); degree of self-sustainability of the restored ecosystem (Lugo 1992, SER 2004); analysis of trajectories by establishing trends from periodic assessments of the restored area (Zedler and Callaway 1999); and comparative functional analysis of restored systems (Tongway and Hindley 1995, 2004). Most of these approaches can be grouped into one of the following main types: (1) measuring the achievement of specific goals and stages, (2) direct comparison with reference sites or between restoration alternatives, and (3) assessment of ecosystem quality. The three categories partially overlap, as both restoration goals and quality indicators are commonly defined in relation to some sort of reference. In practice, there are particular pros and cons associated with the implementation of each of these evaluation approaches (see below).

Achievement of restoration goals and evaluation

Perhaps the most obvious evaluation approach is to measure the degree of achievement of the proposed objectives. Indeed, it is well-established in the literature that evaluation criteria need to relate back to specific restoration goals and explicit expectations (e.g., Aronson et al. 1993, Toth and Anderson 1998, Hobbs and Harris 2001). Ideally, based on the general goals of the restoration project and on the knowledge and understanding of the ecology of the system, explicit predictions are made of expected responses by biotic and abiotic ecosystem components that will then be monitored for evaluation (see Chapter 5, this volume); in turn, designing the appropriate monitoring and evaluation program helps refine and explicitly state the project specific objectives. In some cases, expectations could be written as statements of testable hypotheses, so that evaluation could simply be based on testing one or more null hypotheses (Thayer et al. 2003). However, less clearly defined objectives are more common for most restoration efforts.

Poorly-defined objectives for evaluation may result from our limited understanding of the processes and factors, as well as the biotic interactions and assemblages that control ecosystem dynamics, which in turn limits the definition of the specific outcomes that could be expected from the restoration actions implemented. On the other hand, much of the recent scientific evidence suggests that ecosystems do not always undergo predictable and more or less gradual trajectories (Westoby et al. 1989, Zedler and Callaway 1999). Indeed, ecosystems can exhibit threshold dynamics, change between alternative metastable states, or suddenly develop in an entirely new direction (Hobbs and Norton 1996, Scheffer and Carpenter 2003, Rietkerk et al. 2004, Suding et al. 2004, Bestelmeyer 2006, Suding and Hobbs 2009). Restoration projects may therefore result in a wide range of potential outcomes, some of them being quite unpredictable. Several studies have suggested different approaches that take into account the uncertainty in space and time about restoration outcomes. These include establishing ranges of variability for target attributes, or a range of different potential targets that would be acceptable (e.g., White and Walker 1997, Allen et al. 2002, Palmer et al. 2006); setting goals that recognize multiple end points, considering new models of ecosystem dynamics (Suding and Hobbs 2009); and, in all cases, setting realistic expectations acknowledging the rather unpredictable nature of ecosystem dynamics and the possibility of multiple trajectories (Palmer et al. 2006, Choi 2004). The fact that restored ecosystems are not static also points to the need for establishing the suitable time frame in which to assess the achievement of the various stages envisioned. Depending on the specific type of project, defining several phases and associated goals may be appropriate (Aronson and Vallejo 2006).

Both the social context and the knowledge framework are dynamic, and each influences restoration decisions and objectives. Social values play an important role in defining restoration goals (Diamond 1987, Davis and Slobodkin 2004, SER 2004). Changing socio-economic conditions and new environmental problems can alter the social demands placed on wildlands and, accordingly, new restoration goals emerge. For example, since the 1990s, mitigating climate change has become a core objective of afforestation and reforestation programs worldwide. In the past, the main objectives of reforestation projects in the northern Mediterranean were wood production, soil protection from erosion, and flood control (Vallejo et al. 2006, Bautista et al. 2010); while in the last decades the objectives have shifted to other ecosystems goods and services of perceived socio-economic and ecological benefit, such as improvement of water quality, recreation, improvement of wildlife habitats, fire prevention, biodiversity conservation, etc. Many projects that could be considered as highly successful in meeting originally established objectives, would meet none or very few of the current social demands regarding biodiversity conservation and ecosystem services.

Reference systems for restoration evaluation

Restoration ecologists usually advocate the use of target or model communities as reference systems to set restoration goals and evaluate restoration success (e.g., Aronson et al. 1993, Aronson and Le Floc'h 1996, Brinson and Rheinhardt 1996, White and Walker 1997, Ruiz-Jaen and Aide 2005). This idea, which is also stated by the SER Primer on Ecological Restoration (SER 2004), has been embraced by a number of restoration monitoring guidelines produced by environmental agencies (see, for example, Davis and Muhlberg 2002, Thayer et al. 2003). A reference system is any ecosystem or landscape showing the structure and function that is expected for an area to be deemed successfully restored. Given natural variability, some authors suggest the assumption of variation in the selected reference system, incorporating information from diverse sources extending across the ranges of ecological variation possible (e.g., White and Walker 1997, Allen et al. 2002).

Reference conditions are commonly defined in terms of compositional and structural elements. A restoration process aimed at reconstructing a prior ecosystem and re-establishing former communities is, however, a very difficult task, particularly at the landscape level (Henry and Amoros 1995, Hobbs and Norton 1996, Bradshaw 1997, van Diggelen et al. 2001). Several authors have called for an alterative approach based on evaluation criteria that focus on the functional aspects of the reference system, using specific services or certain functions as reference conditions (e.g., Brinson and Rheinhardt 1996, Choi 2004). Falk (2006) proposed to replace the more static concept of reference conditions by reference dynamics: a process-centered approach that places emphasis on ecological functions and ecosystem processes.

Historical data on pre-disturbed conditions or remnants of historic natural areas are common forms of target references (Holl and Cairns 2002, Hobbs and Harris 2001). However, candidates for natural reference areas in the Mediterranean basin, after centuries of land use and degradation, are very scarce (Vallauri et al. 2002, Aronson and Vallejo 2006). Moreover, several studies point to the usefulness of using historical data as reference information given the dynamic nature of communities in a changing environment and socio-economic context (Pickett and Parker 1994, Hobbs and Norton 1996, Choi 2004). Without denying that success stories exist, Hilderbrand et al. (2005) pointed out that much of the field evidence does not support that restored ecosystems will return to their pre-disturbed state, and warned against this assumption as it is used to justify exploitation of natural resources in undisturbed environments. Zedler and Callaway (1999) reported that few created or restored wetlands achieved structure or function equivalent to existing wetlands. Similarly, a recent meta-analysis review of 89 restoration assessments by Rey Benayas et al. (2009) reported that ecological restoration increased provision of biodiversity and ecosystem services by 44 and 25%, respectively. However, values of both remained lower in restored versus intact reference ecosystems, at least in decadal time scales.

Some studies suggest that rather than focus on restoring to some primeval state, a more profitable approach for restoration would be to focus on repairing damaged systems to the extent possible, considering both the ecological potential for restoration and societal desires (Higgs 1997, Hobbs and Harris 2001). In this approach, the pre-restored, degraded system can be considered as the reference with which to evaluate restoration. Defining the degree of improvement that could be considered a success is the particular challenge of this approach. Some degraded systems have shifted to a new state that is reinforced by internal feedbacks and cannot be restored to the previous state unless certain thresholds are passed (Whisenant 1999, Suding et al. 2004). Knowledge about these restoration thresholds is still very scarce (Maestre et al. 2006). Furthermore, due to the many interactions involved, a single predictive threshold value seems unlikely to emerge (Bestelmeyer 2006), which limits the definition of reference target values for evaluation.

Finally, evaluation can be centred on the comparison of restoration alternatives. This approach does not rule out including either intact reference systems or degraded pre-restored systems within the set of compared cases. Methods for generic functional analyses (e.g., Tongway and Hindley 2004, Herrick et al. 2005) and cost-benefit analyses (Macmillan et al. 1998, Kirk et al. 2004) are of particular interest for comparative restoration evaluation, as they provide indices that can be directly comparable across restoration sites differing in area or scale.

Whatever the references or restoration alternatives used for comparison, the selection of the variables to be assessed is key to the evaluation process. The structural and functional attributes of ecosystems do not always linearly covary, nor do the environmental and socio-economic impacts and constraints of restoration actions (Cortina et al. 2006, Rey Benayas et al. 2009). Therefore, when comparing between the restored area and the selected references and/or alternatives, results may vary greatly depending on the variables considered.

Evaluation as quality assessment

This approach is related to existing tools and methods for ecosystem monitoring and assessment, which typically consider a wide set of attributes to evaluate ecosystem status and integrity. For example, WWF-World Wide Fund for Nature and IUCN-International Union for Conservation of Nature have developed an approach to landscape assessment of forest quality that can also be used to evaluate restored forests. This method is based on the following criteria: (1) Authenticity - including composition, pattern, functions, processes, and management practices; (2) Forest health - including health of trees and other forest flora and of fauna, and robustness to changing environmental conditions; (3) Environmental benefits - including biodiversity and genetic resource conservation, and soil and watershed protection; (4) Social and cultural values - including wood and non-timber products, employment and subsistence, recreation, and historical, cultural, aesthetic and educational values (Dudley et al. 2006).

The SER Primer (SER 2004) provides a list of nine ecosystem attributes as a guideline for measuring restoration success. The first attribute bases success on the similarity between the restored area and the reference sites, while the rest of the attributes can be considered as quality indicators (e.g., presence of indigenous species; presence of functional groups necessary for long-term stability; integration with the landscape; resilience to natural disturbances; self-sustainability) that focus on the actual condition of the restored area regardless comparisons with reference sites. Some of these attributes are perhaps too generic for being directly assessed and must be viewed as framework criteria for developing specific quantitative indicators.

In this approach, the restored area is assessed through a variety of quality indicators that reflect current social demands, yet they may not be the original target attributes considered by the restoration project. When no real, existing reference is available, or when there is no adequate information about pre-restored conditions, this approach can be the appropriate framework for evaluation.

Evaluation as information systems

All the approaches above implicitly consider restoration evaluation as the evaluation of restoration success (Hobbs and Harris 2001). However, restoration success is a subjective and somehow unclear and elusive concept (Zedler 2007) that does not fully recognize and accommodate the many potential sources of variation and uncertainty concerning restoration outcomes, such as, for example, existing knowledge gaps on the ecological theory that supports the selected restoration strategies; the inherent uncertainty associated with both the on-the-ground implementation of restoration projects and the natural dynamism of the restored areas; the tradeoffs between ecosystem services; or the diverse, even contrasting perspectives among the various stakeholders. There are many contexts where measuring success is nevertheless feasible and appropriate. Thus, there are cases in which objectives or reference values for the attributes of interest, as well as their acceptable range of variability, are well defined and hence the meaning of *success* is clarified.

Rather than merely following a success *vs* failure approach, evaluation may be viewed as a process of creating information and knowledge on the restoration actions implemented, providing a more or less comprehensive and multifaceted description of the restoration outcomes. This approach considers evaluation as an information system that collects and provides useful data on ecosystem and landscape responses to restoration. It can therefore support any other approach to evaluation. According to this view, evaluation should rely on the widest range possible of attributes and perspectives, provided they are relevant (see the REACTION protocol below). The challenge for this approach is to organize and integrate the profuse information in a harmonized way, allowing conclusions to be drawn, avoiding redundant information, and keeping the number of attributes assessed manageable in practical terms.

What to evaluate? Selection of attributes and indicators[1] for evaluation

A large number of qualitative and quantitative variables can be used to evaluate a restored ecosystem. Since the choices may affect the interpretation of restoration outcomes, the selection of variables for evaluation is often a thorny issue to address. Evaluation criteria have evolved parallel to changes in conceptual frameworks and perspectives for restoration, which have in turn been reflected in the type of attributes and indicators selected for monitoring and assessment. Thus, traditional evaluation approaches that commonly focused on technical aspects of compliance success (e.g., seedling survival rates in forest plantations; Alloza 2003),

1. Here the term indicator refers to any biophysical or socio-economic variable or statistic used to assess land condition.

have given way to approaches that also include structural and functional indicators of ecosystem health and integrity (Xu et al. 2001, Ruiz-Jaen and Aide 2005). Moreover, given the profound connections existing between the ecological and the socio-economic systems (Turner et al. 2003, Liu et al. 2007), it is increasingly recognised that the assessment of land condition must be based on both biophysical and socio-economic attributes (MEA 2005), which also applies to the evaluation of restoration actions (SER 2004, Zucca et al. 2009).

Irrespective of the biophysical or socio-economic attributes assessed, the selected indicators should be relevant, be sensitive to variations of environmental stress, respond to stress in a predictable and scientifically justifiable manner, but also be simple and measurable with a reasonable level of effort and cost (Dale and Beyeler 2001, Jorgersen et al. 2005). Because of the large spatial and temporal variability of ecosystems, particularly in drylands, recent studies suggest focusing ecosystem assessment on 'slow' variables (Carpenter and Turner 2000), both biophysical and socio-economic (e.g., soil fertility, market access), as high variability in 'fast' variables may mask fundamental trends and long-term changes (Reynolds et al. 2007). Similarly, the variables used should have low spatial variability – outside of recognised gradients.

In defining an evaluation approach and selecting the appropriate indicators, there are a number of key aspects to consider that concern the number of indicators, their spatial and temporal scope of application, and the scale, methods, and resolution of the measurements. Obviously, the options chosen largely depend on the objectives of the restoration project, but also on the conceptual framework that underlies the evaluation approach to be followed. Evaluation approaches based on few site-specific indicators would fit restoration projects with relatively straightforward, project-specific objectives or projects that are applied to a particular and relatively small piece of land over a limited period of time. For example, the local recovery of the population of certain species is often the primary goal of restoration, due either to ecological, economic or cultural reasons. The achievement of such a specific goal can be assessed through the monitoring of few indicators related to the dynamics and sustainability of the target population (Bash and Ryan 2002). Similarly, if the objective in a restoration project is to reduce the abundance of an invasive species and enhance the performance of a number of target native species, then evaluation should measure the abundance (e.g., cover, biomass) of the invasive species and the response (e.g., seedling survival, cover, growth, etc.) of the native species before and after treatment (Hartman and McCarthy 2004). However, the selection of appropriate indicators is not so straightforward for projects with more general goals, or for projects that apply to a relatively broad geographic area (i.e., to the landscape scale).

The ultimate goal of many ecological restoration projects is to recover ecosystem health and integrity, to return ecosystem structures, functions, and processes to reference conditions, and/or to enhance the provision of ecosystem goods and services (SER 2004, Blignaut and Aronson 2008). Which metrics should be used to evaluate projects with such general objectives? There is no universal prescription for what to measure in order to describe the ecosystem response to ecological restoration. In general, the spectrum of alternatives ranges from project-specific options to more broadly applicable selection of indicators, and from simple indicators of ecosystem integrity to indicator suites of a large variety of attributes (Fig. 2). In making these choices of evaluation approaches and indicators, there are several trade-offs to consider. On the one hand, the use of single indicators of ecosystem integrity can be a very cost-effective option that reduces monitoring effort, but it requires a profound knowledge of how well the indicator represents the structural and functional conditions expected for the restored system. Furthermore, the loss of information when many variables are integrated into a single index could mask real differences between management and restoration options. On the other hand, the more site- or project-specific the indicators, the more useful the resulting information for local managers to adjust restoration practice within an adaptive management framework. However, the evaluation results of such a tailored approach would apply only to the site and conditions under study, hindering the applicability to broad geographic areas and the comparison of restoration strategies across a variety of sites and regions.

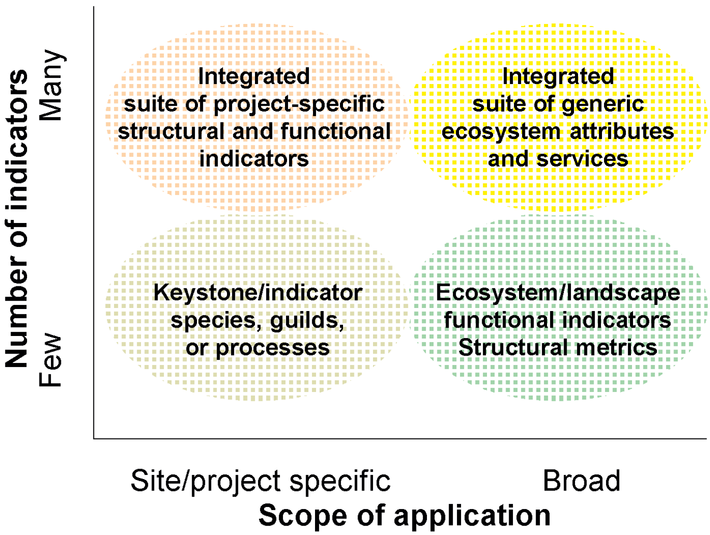

FIGURE 2. General range of alternatives for ecological evaluation of restoration projects as defined by the number of indicators and their scope of application.

Evaluation approaches based on few holistic indicators

Simplification towards essential indicators that could characterise ecosystem recovery adequately is obviously a cost-effective approach to evaluation. For example, measures concerning indicator species, umbrella species, guilds, or assemblages of indicator species are often used as surrogates of ecosystem function and integrity (e.g., Williams 1993, Patten 1997). The structural and functional requirements of indicator species should reflect the conditions expected in the restored ecological system. This approach requires the development of a conceptual model that outlines the structure of the community, including interrelations among ecosystem components (Block et al. 2001). Therefore, the accurate use of these indicators depends on a high level of knowledge of the target system. Although evaluation protocols based on indicator species are relatively site-specific, when based on general taxa or guilds (e.g., bird populations, biological crusts) they could be applied to a broad range of sites and project types (Neckles et al. 2002, Bowker et al. 2006).

General indicators of community structure, such as species richness, diversity, and evenness, can also be used for evaluating general ecosystem response to restoration (e.g., Reay and Norton 1999, Passell 2000, Gómez-Aparicio et al. 2009). These structural indicators may or may not accurately reflect the recovery of ecosystem function (Ryder and Miller 2005), and, of course, do not take into account species identities and their potential role as keystone species, noxious weeds, or any other particular role played by single species or functional groups. However, recent results indicate that biodiversity is positively related to the ecological functions that support the provision of ecosystem services in restored areas (Rey Benayas et al. 2009). Biodiversity assessments typically focus on particular biota, ranging from general groups (e.g., plants, vertebrates, herbaceous species) to specific taxa or guilds (e.g., butterflies, resprouting shrubs), often resulting in a combined approach based on biodiversity and indicator taxa (Kerr et al. 2000).

Vegetation cover and composition are the most common metrics used for evaluating restoration projects, as it is often assumed that the recovery of fauna and ecological processes will follow the establishment of vegetation (Ruiz-Jaen and Aide 2005). Since vegetation cover is relatively easy to assess, it is commonly used as a surrogate of ecosystem functions and habitat quality (Reay and Norton 1999, Robichaud et al. 2000, Wilkins et al. 2003, Wildham et al. 2004). However, vegetation cover alone cannot always reflect how well an ecosystem is functioning. For example, a number of studies in semiarid areas have shown that shape, spatial orientation and arrangement of plant patches within a landscape greatly influence hydrological functioning (e.g., Ludwig et al. 1999, Puigdefábregas 2005, Bautista et al. 2007).

During the last decade, a variety of functional assessment approaches that assume a tied relationship between semiarid ecosystem functioning and the spatial pattern of vegetation have been proposed. The theoretical framework for these approaches considers that landscapes occur along a continuum of functionality from highly patchy systems that conserve all resources to those that have no patches and leak all resources (Ludwig and Tongway 2000). Some of the functional assessment methods are based exclusively on single vegetation/soil pattern attributes (Bastin et al., 2002; Ludwig et al., 2007; Kéfi et al. 2007, Mayor et al. 2008), while others also incorporate properties relative to the soil surface condition (Tongway and Hindley, 2004; Herrick et al., 2005). For example, the “Landscape Functional Analysis” (LFA) methodology (Tongway and Hindley, 1995, 2004) assesses ecosystem functional status through a set of easily recognizable soil and landscape features, from which indices of infiltration, stability and nutrient cycling are derived. These indices are expected to reflect the status of water conservation, soil conservation, and nutrient cycling processes in the target ecosystems.

Evaluation approaches based on large suites of indicators

On the other end of the spectrum of options (Fig. 2) is evaluation based on a relatively large suite of indicators aimed to present a more comprehensive diagnosis of the restoration effects. An integrated suite of indicators may be specific for certain sites, problems or project types (Keddy and Drummond 1996, Davis and Muhlberg 2002, Palmer et al. 2005) or may mostly rely on general metrics that can be used for assessing a wide range of cases. Several authors have promoted approaches that combine both general and case-specific indicators (e.g., Neckles et al. 2002, Jorgersen et al. 2005). The use of multiple indicators maximizes the amount and variety of information provided on the restored area and is the best approach possible when there is not sufficient scientific knowledge to support the use of single holistic indicators as proxies for the function and integrity of the target ecosystem.

Numerous authors have proposed lists of attributes that can be used as conceptual frameworks for designing ecological restoration projects and evaluating restoration success (e.g., (Ewel 1987, Aronson and Le Floc'h 1996, Hobbs and Norton 1996, SER 2004, Palmer et al. 2005). The SER Primer (SER 2004) proposed a list of nine attributes that includes diversity and other structural properties (such as presence of indigenous species and presence of functional groups necessary for long-term stability), and general ecosystem functions (such as resilience to natural disturbances and self-sustainability). Ruiz-Jaen and Aide (2005) supported the use of the SER attributes, but promoted a simplified framework that considers three main categories: diversity, vegetation structure, and ecological processes. Although these attributes and categories provide a useful basis for guiding the selection of indicators for evaluation, they need further specification to be readily assessed through site-specific criteria (Choi 2004). There has been a greater emphasis on biophysical criteria for evaluating the outcomes of restoration efforts, while socio-economic indicators are less addressed. Because of its focus on provision of services is directly linked to human well-being, the conceptual development by the Millennium Ecosystem Assessment (MEA 2005) has provided a robust integrated framework for evaluation. Nevertheless, there still is a great need for practicable methodologies that integrate biophysical, socio-economic, and cultural indicators (See REACTION approach below).

Scale and resolution for indicator assessment

To address the widest scope of restoration effects on ecosystems, landscapes, and society, as well as their cross-scale interactions, a multiscale approach to evaluation is always advisable. For example, regarding forest restoration projects, a stand- or site-scale assessment may focus on technical aspects, structural and functional ecosystem attributes, and on a market-based economic valuation perspective (prized goods and services), while landscape- and regional-level indicators would describe general impacts on the environment and public/social welfare (Fig. 3). Similarly, short-term evaluation may rely on technical and ecological indicators that communicate implementation and compliance success, allow for predicting the likelihood that a function is occurring, help identify problems, and guide

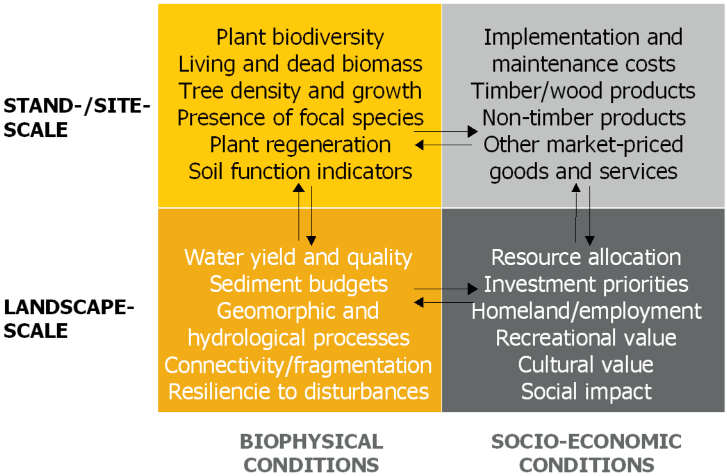

FIGURE 3. Example of a multi-scale integrated framework and indicators for evaluating forest restoration projects.

adaptive management. For example, early assessment of seedling survival and growth in reforestation projects can help predict the likelihood that the desired forest structure is eventually achieved, and allow for corrective actions if needed (see Chapter 5). However, most restoration projects require a number of years for some expected processes and dynamics to take place and, therefore, goal achievement and general structural and functional quality of the restored areas should be evaluated using long-term assessment data.

As with any ecological study, the choice of spatial and temporal resolution for the monitoring and evaluation depends on the variables and questions being addressed (White and Walker 1997, Block et al. 2001), but also on the decisions made by the practitioners regarding the trade-off between the effort needed and the information provided. Assessment methods range from simple, qualitative assessments based on field observations (e.g., a high-medium-low ranking system, photo-points, visual estimates) to relatively complex protocols based on quantitative measurements of critical ecosystem attributes (Machmer and Steeger 2002). Regarding the time frame for monitoring, assessment methods range from single-time assessment to continued observations designed to assess trajectories and account for the interannual variability of ecosystem functions.

The relatively recent extraordinary development and accessibility of products from global and regional scale remote-sensing (RS) systems have led some international bodies to recommend the integrated use of RS-based geospatial information with ground-based observations to assess vegetation and soil condition (MEA 2005, ICCD/COP8/CST 2007). Indeed, there is an increasing use of RS technology to trace land condition at the landscape scale (Díaz-Delgado et al. 2002, Roder et al. 2008, van Leeuwen et al. 2010), though its utility for assessing the efficiency of restoration actions remains limited (van Leeuwen 2008).

An integrated protocol for the evaluation of forest restoration in the northern Mediterranean: The REACTION approach

Since the late-19th century, and particularly during the first half of the 20th century, significant national-scale attempts to restore degraded drylands were implemented in the northern Mediterranean countries. These efforts were mostly based on large afforestation and reforestation programs (see Chapter 1). In many cases, the restoration strategy relied on the introduction of fast-growing pioneer species, with the assumption that these species would then facilitate the introduction of late-successional hardwoods (Pausas et al. 2004). The main species planted were native pines, such as *Pinus brutia*, *P. halepensis*, *P. nigra*, etc., though exotic species also were planted. As a whole, these large-scale reforestation programs constitute an impressive testing ground for assessing restoration strategies and techniques. However, most reforestation actions were not followed up with subsequent monitoring and the results obtained have rarely been assessed (Gómez-Aparicio et al. 2009). Since the real outcome of a reforestation project can only be evaluated comprehensively in the long term, i.e., after several decades, the projects implemented during the 19th and 20th centuries offer a unique opportunity to assess the potential of reforestations as tools for restoring Mediterranean forests. Acknowledging this opportunity, the REACTION project (Restoration Actions to Combat Desertification in the Northern Mediterranean)[2], has recently developed an integrated approach to evaluate forest and woodland restoration actions in the northern Mediterranean. The REACTION evaluation protocol (http://www.gva.es/ceam/reaction) was not only conceived as an evaluation methodology but also as an information system designed to compile and disseminate the information derived from the restoration projects evaluated.

The evaluation of old reforestation projects entails major difficulties such as the lack of monitoring data, the lack of reference sites, and the highly heterogeneous, and often very scarce, information available about project goals and implementation. In addition, the originally established goals commonly meet none or few of the current social demands regarding ecosystem services. To address these challenges, the REACTION approach combines three main evaluation criteria: (1) degree of achievement of specific initial project objectives, (2) comparative analysis between pre-restoration degraded conditions and current conditions, and (3) analysis of current quality of the restored system irrespective of initial project goals. Furthermore, the REACTION protocol has been designed as a broad framework that uses a wide variety of indicators, optimising the use of existing available information and requiring minimum field assessment. The selected indicators relate to ecosystem integrity and services, and to socio-economic and cultural attributes that are relevant for Mediterranean conditions.

The REACTION protocol includes eight sections (Table 1). Sections I to IV provide context information on the site and the restoration project, while sections V to VII address the

2. REACTION was funded by the European Commission under the Fifth Research, Technology, and Development Framework Programme, and involved research groups and forest managers from Freece, Italy, France, Portugal, and Spain.

evaluation of the restored area. Most of protocol considers a landscape perspective for evaluation, as many of the expected biophysical and socio-economic impacts of forest restoration projects appear at the landscape scale. However, sections IV and V compile context and evaluation data for any single restoration unit[3] or stand included within the project, and are meant to be replicated as many times as the number of restoration units in the project.

To allow for analysis of the conditions and technical approaches that influence restoration outcomes, data on the environmental and socio-economic context and on the

TABLE 1. General structure of the REACTION evaluation protocol.

I. GENERAL INFORMATION	1. GENERAL DESCRIPTION 2. DATA SOURCES
II. SITE DESCRIPTION	1. CLIMATE 2. TOPOGRAPHY 3. GEOLOGY 4. SOILS 5. ECOLOGY 6. DEGRADATION IMPACTS AND DRIVERS
III. RESTORATION PROCESS	1. GOALS 2. PLANNING 3. COST AND FINANCING 4. GENERAL TECHNICAL DESCRIPTION 5. MONITORING AND ASSESSMENT 6. ENVIRONMENTAL OR TECHNICAL UNITS
IV. TECHNICAL DESCRIPTION BY RESTORATION UNITS	1. UNIT DESCRIPTION 2. SPECIFIC ENVIRONMENTAL CHARACTERISTICS 3. PROMOTION OF AUTOGENIC RESTORATION 4. PRIOR ACTION ON BRUSH VEGETATION 5. SITE PREPARATION 6. PLANTING AND SEEDING 7. FIELD TREATMENTS/MAINTENANCE WORKS/ MANAGEMENT
V. ASSESSMENT BY RESTORATION UNITS	1. PLANTATION/SEEDING RESULTS 2. STRUCTURE AND BIODIVERSITY 3. FUNCTIONS AND PROCESSES 4. STAND/UNIT HEALTH
VI. PROJECT ASSESSMENT	1.LANDSCAPE AND ENVIRONMENTAL ASSESSMENT 2.SOCIO-ECONOMIC ASSESSMENT
VII. EVALUATION SUMMARY	
VIII. EXPERT JUDGEMENT	

3. Restoration unit refers to any area or stand within the restoration project area that present particular enviromental (e. g., microclimate, geology, soil type) or technical (e. g., treatment applied, implementation date) characteristics.

technical characteristics of the restoration project are core to the evaluation protocol. Thus, section II of the evaluation protocol describes the climate, topography, geology, soils, and ecosystems, as well as the main degradation impacts and drivers in the target restoration area. Section III organizes the available information on project design and implementation through a set of questions about goals, planning, financing, and other technical details (Table 2). Finally, section IV describes specific environmental characteristics and technical details of the restoration action for each stand or landscape unit within the restoration project.

The assessment of the restoration units and/or forest stands within the restored area provides information on plantation/seeding results, ecosystem structure and diversity, ecosystem functions and processes, and stand health (Table 3). This biophysical evaluation focuses on the current quality of the restored ecosystems, taking into account recent advances in indicators for land quality assessment (see, for example, WWF 2002). The structural quality of the restored area is measured through a number of biodiversity, key species and spatial pattern indicators. The functional evaluation relies on indicators that reflect hydrological and nutrient cycling processes, as they are particularly relevant for the conservation of limiting resources in Mediterranean degraded and desertification-prone lands.

Project evaluation at the landscape level encompasses both biophysical and socio-economic assessment (Table 1). Landscape and environmental assessment provides information on the distribution of ecosystem types in the area; the presence and types of protected areas; landscape pattern (habitat connectivity/fragmentation); visual impacts; and flooding and erosion assessment at the catchment/landscape scale as compared with pre-restoration conditions. The socio-economic assessment focuses on information about land use, ecosystem goods and services, employment, and the recreational, educational, and cultural values of the restored land (Table 4).

Finally, section VII summarizes the project evaluation by grouping the information provided by the large variety of indicators considered in the previous sections into a small suite of categories that represent ecosystem structure and services (Table 5). This final summary contributes to the standardization of project evaluation, facilitating comparisons among projects and context conditions.

In addition to assessing the restoration projects through the various sections and indicators described above, the REACTION protocol includes a process where expert overall judgments of both natural resource managers and researchers involved in the evaluation of the restoration project are obtained. This provides insights not readily available in the assessment of the data and facilitates the engagement between researchers and managers.

Major innovations of the REACTION protocol are the large amount of detailed information compiled on well-documented restoration projects; the integrated approach to evaluation, and the regional (Mediterranean) scope. The REACTION evaluation methodology has been applied to 40 forest restoration projects implemented in Greece, Italy, France, Spain, and Portugal, ranging in size from ~100 to 3,500 ha. The projects aimed mostly to restore pine forests and mixed pine-oak forests and are representative examples of the varied approaches to forest restoration in the northern Mediterranean. A key outcome of the REACTION project was the Database for Mediterranean Restoration Projects (http://www.ceam.es/reaction), an open-access database that includes the projects compiled and evaluated.

TABLE 2. Questionnaire for the description of the design and implementation characteristics of a restoration project (section III in the REACTION evaluation protocol).

III. RESTORATION PROCESS (DESIGN AND IMPLEMENTATION)
III.1. RESTORATION GOALS
1. What were the general project's defined objectives?
2. Project scope: ☐ Restoration programme ☐ Pilot project ☐ Research ☐ Educational ☐ Other
3. Structure goals: a) Target biological communities/ecosystems to be restored: b) Does this project target the protection or conservation of specific species? *Species*: c) Does this project try to introduce a species as part of restoration or conservation efforts?. List species: d) Does this project try to eradicate a species as part of restoration or conservation efforts?. List species: e) Other structural goals (e.g., promote understorey cover, increase growth, establish forest mosaic, etc):
4. Functional goals and expected ecosystem services: ☐ Agriculture production ☐ Biodiversity conservation ☐ Fire control ☐ Air quality ☐ Forestry production ☐ Wildlife habitat ☐ Weed control ☐ CO_2 sink ☐ Grazing/pasture lands ☐ Flood control ☐ Seed source ☐ Other: ☐ Hunting ☐ Erosion control ☐ Water quality
5. Goals at landscape level: ☐ Increase connectivity ☐ Increase landscape diversity ☐ Increase forest surface ☐ Rural planning ☐ Other:
6. Which ecosystem goods were expected to be obtained/increased? ☐ Wood products: ☐ Animal products: ☐ Non-timber forest products (e.g., edible mushrooms, aromatic plants, etc.): ☐ Others:
7. Was the enhancement of the recreational/tourist/cultural value of the area a specific goal? If yes, provide details:
8. Was the creation of jobs a specific goal? If yes, specify number and type (permanent/seasonal):
III.2. PLANNING
1. Main stages provided for the control/reduction of degradation causes (passive restoration)? When and for how long were these stages to be carried out?
2. Main stages provided for the field work (active restoration)? When and for how long were these stages to be carried out?
3. Does the forest have a management plan? If yes, what is the legal/policy framework?
III.3. COST, FINANCING, AND PARTICIPANTS
1. Project cost and financing a) Total cost of implementation (Euros): Ref. Date: c) Sources of financing: b) Average annual cost for maintenance (Euros): Ref. Date: d) Other relevant information:
2. International, national or regional programmes and/or plans related to the project:
3. Agencies/groups involved in the project:
III.4. GENERAL TECHNICAL DESCRIPTION
1. Structures and facilities developed (if any) within the project. Describe:
2. Has any traditional technology been applied? Describe:
3. Has any innovative technology been applied? Describe:
4. Are there any quality standards for the seedlings? Describe:
5. Are there quality standards for the work? Describe:
6. Are there defined success criteria for the project? Describe:
III.5. MONITORING AND ASSESSMENT
1. Was there any monitoring/assessment carried out? If yes, a) Which elements were taken into account?: ☐ Technical ☐ Ecological ☐ Socio-economic b) At what intervals/period was the monitoring carried out? c) Describe briefly the monitoring methodology:

TABLE 3. Questionnaire for the technical and ecological assessment of restoration projects by means of the REACTION protocol.

V. ASSESSMENT BY RESTORATION UNITS*

1. Project acronym/code (see I.11b):	**2. Unit number/code** (see IV.1.2):	**3. Reference (assessment) date:**

V.1. PLANTATION/SEEDING RESULTS

Indicate the average or the most representative data for each variable (when applicable) and species.

1. Have the performance standards for the seeds/seedlings been attained? *(Only if there were quality standards)*
☐ Yes ☐ No ☐ Partly ☐ Unknown

2. Have the performance standards been attained for the work (site preparation, plantation,...)?
☐ Yes ☐ No ☐ Partly ☐ Unknown

3. Plant cover (%)	Total:	Tree species:	Shrub species:	Herbaceous species:
4. Above-ground biomass (kg/ha)	Total:	Tree species:	Shrub species:	Herbaceous species:

5. Species used For each species used, indicate:	by planting	by seeding	a. Survival (%)	b. Density (individuals/ha)	c. Cover (%)	d. Height (m)	e. Diameter at breast height (cm)	f. Basal diameter (cm) *(only for young plants)*	g. Wood volume (m3/ha)	h. Natural regeneration? (yes/no)	i. Average age (years)	j. Age distribution		
												Mixed – young	Mixed – old	Mono
	☐	☐										☐	☐	☐

V.2. STRUCTURE & BIODIVERSITY

1. Stand/unit age: ☐ Old ☐ Mature ☐ Young

2. Tree canopy structure: ☐ Multi-layered ☐ Mono-layered ☐ Absence of tree layer

3. Understorey: ☐ Varied and multi-layered ☐ Herbaceous layer and scatter woody plants ☐ Absent ☐ Other
Describe:

4. Spatial distribution of trees: ☐ Regular ☐ Slightly clumped ☐ Patches and gaps

5. How natural is the composition of tree species? ☐ Fully ☐ Partly ☐ Exotic

6. How natural is the composition of other species? ☐ Fully ☐ Partly ☐ Exotic

7. List the alien species present, if any:

8. Cover of resprouter species: Total: Tree species: Shrub species:

9. Biological inventories: *Please list species inventoried and indicate number of species by taxa*, if available

Taxa	**a) Before the project. Reference date:**	**b) After the project. Reference date:**
For each taxa, indicate:	Inventory:	Inventory:
	Dominant species:	Dominant species:
	Rare / Endangered / Threatened / Protected:	Rare / Endangered / Threatened / Protected:
	Total number of species inventoried:	Total number of species inventoried:

10. Absence of keystone or dominant species that would be expected: ☐ Yes ☐ No ☐ Unknown
If yes, please list:

11. Are any functional groups (shrub layer, annual legumes, perennial grasses, etc.) missing or endangered?
☐ Yes ☐ No ☐ Unknown. If yes, please list:

12. Presence of key species indicative of ecosystems at particular succesional stage: ☐ Yes ☐ No ☐ Unknown
If yes, please list:

13. Presence of key species indicative of integrity of food webs: ☐ Yes ☐ No ☐ Unknown
If yes, please list:

14. Are there any genetic data available: ☐ Yes ☐ No ☐ Unknown
If yes, please list:

* This section is meant to be applied to each restoration unit in the restored area.

TABLE 3 (cont.). Questionnaire for the technical and ecological assessment of restoration projects by means of the REACTION protocol.

V. ASSESSMENT BY RESTORATION UNITS					
1. Project acronym/code (see I.11b)**:**		**2. Unit number/code** (see IV.1.2)**:**		**3. Reference (assessment) date:**	
V.3. FUNCTIONS & PROCESSES					
1. Are there significant amounts of dead wood present in varying stages of decomposition? ☐ Snags ☐ Down logs ☐ Not significant					
2. Average organic horizon thickness (cm):					
3. Soil surface conditions: Bare soil (%): Degree of soil sealing/crusting: ☐ None ☐ Slight ☐ Moderate ☐ Severe Presence of significant patchy or continuous biological crust? ☐ Yes ☐ No					
4. Erosion/accumulation type ☐ None ☐ Sheet erosion ☐ Rill erosion ☐ Gully erosion ☐ Badlands ☐ Accumulation ☐ Wind erosion/deposition ☐ Others (describe):			**5. Erosion/accumulation intensity** ☐ Nil ☐ Slight ☐ Medium ☐ Moderate ☐ Severe ☐ Extreme		
6. Stand dynamics Succesional dynamics measured or observed (e.g., abandoned crop→gorse shrubland→pine forest→mixed forest): a) Since the project implementation: → → → → b) Of non-restored reference area nearby during the same period: → → → →					
7. Did any relevant disturbance, such as fire, severe drought/frost, floods, pollution event, etc., affect the restored unit since project implementation? ☐ Yes ☐ No ☐ Partly					
8. Disturbance regime and Regeneration pattern (for each major disturbance recorded in the area, indicate)**:** Disturbance type: Date/s (year/s): Autosuccesion (yes/no):			Relevant composition change (yes/no): Relevant land degradation (yes/no): Describe regeneration pattern:		
9. Any available data on stand productivity/carbon sequestration? Describe:					
V.4. STAND/UNIT HEALTH					
1. Are there significant pests, diseases or invasive species? ☐ Yes ☐ No					
2. Are there significant damages caused by abiotic factors? ☐ Yes ☐ No					
3. Species affected	**4. Dead trees/shrubs** (None, Some, Many)	**5. Degree of defoliation** (None, Slight, Moderate, Severe)	**6. Degree of discoloration** (None, Slight, Moderate, Severe)	**7. Main abiotic factor causing the damages**	**8. Main biotic factor causing the damages**

* This section is meant to be applied to each restoration unit in the restored area.

TABLE 4. Questionnaire for the socio-economic assessment of restoration projects by means of the REACTION protocol.

VI.2. SOCIO-ECONOMIC ASSESSMENT
1. What types of exploitation were and are most common in the area? For each type, provide: 1- % of project area before* the project (*Indicate the reference date): 2- % of project area at present: 3- Date of abandonment (if applicable):
2. Does significant grazing take place in the project area? Indicate species and livestock population (data on past, present and projections for future, if available)
3. Are timber and other wood products exploited?: a) Type of timber and other wood products (species): b) Volume produced/year: c) Is timber and other wood products felled for use by local people?. If yes, describe:
4. Are non-timber forest products gathered? a) Products gathered: b) What is their economic importance? (high/medium/low): c) Does hunting take place?
5. Employment a) Did project implementation works generate jobs for the local population? b) Does the restored area provide jobs at present? ☐ Yes ☐ No ☐ Occasional ☐ Permanent. Describe: c) Number (approximate) of people employed in the restored area? Occasional/year:_____ Permanent:_____
6. Homeland a) Are people living in the restored area? Indicate type of lifestyle: ☐ Indigenous ☐ Settled ☐ Part-time/Second home b) Human population dynamics in the project area in the last 20 years: Type (increase/decrease): ____ Rate of change (low/medium/high): ____
7. Recreational and educational value a) Uniqueness of particular sites within the restored area? If yes, describe: b) Do people use the restored area for recreation? c) Average number of visitors/year (*approximate value*): d) Presence of tourist or educational facilities (*visitor centre, guide trails,...*): If yes, list number and types: e) Types of activity (walking, hunting,...) f) Is the area used for scientific work? If yes, describe:
8. Cultural value a) Does the project area have particular significance to local inhabitants? b) Are there important cultural or religious sites present in the project area?: (*World Heritage sites, sacred groves, trees, burial sites, buildings,...*). List sites, types, designations and indicate if they have official protection: c) Presence of culturally important landscapes: *(land management, grazing system,...)*. Describe: d) Are there references in folklore, literature, etc. to the project area? e) After the project implementation, were there any negative impacts to cultural sites/landscapes? Explain briefly: f) Have the cultural sites/landscapes been protected in the framework of the project?
9. Local participation a) In relation to the project, the local population has a position of? ☐ Participation ☐ Indifference ☐ Opposition ☐ Boycott b) Are local people involved in decisions about the project area? c) What is the nature of participation? d) Has a participatory approach been implemented concerning local people's perception of the project? Was it intended to make the population: - more sensitive to risks (wildfires, floods, erosion, etc.)? ☐ Yes ☐ No ☐ Unknown - more aware of the advantages of ecological restoration? ☐ Yes ☐ No ☐ Unknown - other? (Please specify)

TABLE 5. Summary table for the evaluation of restoration projects by means of the REACTION protocol.

VII. SUMMARY*	
VII.1. ACHIEVEMENT OF PROJECT GOALS	
1. Have the defined success criteria been attained? (Yes/No/There were not defined success criteria) -Only for some restored units and/or criteria. Describe:	
2. Have the structural goal(s) been attained? (Yes/No/ Partly/Only for some units). Describe:	
3. Have the functional goal(s) been attained? (Yes/No/Partly). Describe:	
4. Have the landscape goal(s) been attained? (Yes/No/Partly). Describe:	
5. Have socio-economic goals been attained? (Yes/No/Partly). Describe:	
6. According to survival and growth of planted/seeded species, the plantation/seeding success was: (Very high / High / Medium / Low /Very low)	
VII.2. STRUCTURAL QUALITY	
1. How natural is the composition of the restored ecosystem(s)? (Fully/Partly). Explain:	
2. How natural/mature is the structure and pattern of the restored ecosystem(s)? (Fully/Partly). Explain:	
3. Presence of important biodiversity (Yes/Medium/No):	
4. In the restored area, the project has: (increased / decreased / conserved biodiversity)	
VII.3. FUNCTIONAL QUALITY	
1. Ecosystem dynamics:	Does the restored ecosystem regenerate naturally? (Yes / Not fully). Explain: Do natural successional dynamics occur? (Yes / No /Partly). Explain:
2. Overall functioning:	How are the soil characteristics? (Stable / Slightly degraded / Seriously degraded) How is the potential for nutrient cycling? (High / Medium / Low) How is the ecosystem productivity? (High / Medium / Low)
3. How is the overall ecosystem health? ☐ Good *(No relevant pests, diseases, invasive species, or dead/damaged plants by abiotic factors)* ☐ Medium *(Some individuals affected; low severity level)* ☐ Poor *(Relevant pests, diseases, invasive species, or dead/damaged plants by abiotic factors)*	
4. The project significantly increases:	Resistance (e.g., to grazing, pests, fire, drought): ☐ Yes ☐ No ☐ Partly Resilience (e.g., to fire, pests, drought, etc.): ☐ Yes ☐ No ☐ Partly Erosion control: ☐ Yes ☐ No ☐ Partly Flood control: ☐ Yes ☐ No ☐ Partly
VII.4. LANDSCAPE QUALITY	
1. The project significantly increases:	Forest surface: ☐ Yes ☐ No ☐ Slightly Connectivity of formerly isolated populations: ☐ Yes ☐ No ☐ Slightly Integration among forests and other habitats: ☐ Yes ☐ No ☐ Slightly Habitat diversity: ☐ Yes ☐ No ☐ Slightly The protected surface: ☐ Yes ☐ No ☐ Slightly
2. Aesthetic value: ☐ Very high ☐ High ☐ Medium ☐ Low	
VII.5. SOCIO-ECONOMIC BENEFITS	
1. Cultural value:	Does the project area have particular cultural significance to local inhabitants? ☐ Yes ☐ No The project has ☐ increased ☐ decreased ☐ preserved ☐ created ☐ damaged the cultural value of the site Degree of local participation: ☐ High ☐ Medium ☐ Low
2. Has the project generated ecosystem goods for the local population? ☐ Yes ☐ No Amount of timber and non-timber goods provided: ☐ Very high ☐ High ☐ Medium ☐ Low	
3. Has the project enhanced ecosystem services? ☐ Yes ☐ No Describe:	
4. Does the project contribute to fix/support/increase rural population by increasing tourist and recreational value, by direct employment, or by providing homeland? ☐ Yes ☐ No ☐ Slightly	

* The answer of each question is meant to be derived from the information compiled in the respective previous sections and items.

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Monitoring Guidelines for the Implementation of Forest Restoration Projects in Mediterranean Regions

5

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Introduction

It is generally understood that ecological restoration is still young and evolving (Hobbs and Harris 2001, Winterhalder et al. 2004, Clewell and Aronson 2006). However, the restoration of degraded lands has become increasingly important worldwide, leading to growing demand for ecological restoration expertise, practitioners, and eco-technological products, as well as research in restoration ecology (Dobson et al. 1997, Dudley et al. 2005, Young et al. 2005).

In the Mediterranean Basin, forest restoration is a long-standing practice, which experienced a particularly intense period from the end of the 19th century to the mid-20th century (see Chapter 1, this volume). Restoration of degraded or damaged forest ecosystems includes many restoration activities of which planting is almost always a key component (Harrington 1999). Historically, large-scale tree planting activities constituted the sole restoration action in the Mediterranean basin. Nowadays, the restoration of forest lands includes the use of herbaceous, shrub and tree species, as well as activities aimed at enhancing the autogenic restoration of ecosystems (Vallejo et al. 2006), and there is an increasing interest in working toward the exploration of sustainable forest landscape restoration practices (Boyle 1999, Holl et al. 2003).

Forest landscape restoration has been defined as: "a planned process that aims to regain ecological integrity and enhance human well-being in deforested or degraded landscapes" (Dudley et al. 2005). According to this concept, developed by The World Conservation Union (IUCN), The World Wildlife Fund (WWF), and some of their partners, restoration should not try to re-establish the "pristine" forests of the past. Furthermore, restoration projects are dynamic and their inherent uncertainty needs to be managed; resulting in a process rather than a planned product (Fulé et al. 2002, SER 2004, Saint-Laurent 2005, Falk 2006).

An essential part of any restoration project is an effective monitoring system, which allows the status and trends of selected indicators to be measured and helps to identify the corrective actions and modifications needed (Vallauri et al. 2005). Monitoring increases our understanding about ecosystem and landscape response to restoration treatments and thus plays a major role in reducing and managing uncertainties in restoration actions.

In the Mediterranean basin, due to the historical degradation of forest ecosystems and to the current complexity of a situation in which society demands multiple uses for these ecosystems, together with increasing land use pressure and wildfire occurrence, there is a critical need to assess the results of restoration activities and establish standard monitoring practices as part of the restoration efforts. The aim of this chapter is to provide monitoring guidelines to be applied to forest restoration efforts in the Mediterranean. A methodology for improving monitoring procedures in forest restoration is proposed which includes four monitoring phases –Baseline, Implementation, Functional Assessment, and Long-term monitoring– within an adaptive management framework. Additionally, limitations to monitoring in the Mediterranean region are discussed and solutions anticipated. As a frame of reference, two restoration projects in Andalusia (southern Spain) are used as examples of implemented monitoring activities.

Monitoring of forest restoration projects - An adaptive management approach

Restoration monitoring is the systematic collection and analysis of data that provide useful information for measuring project performance at a variety of scales. It is designed and conducted to provide useful data to understand why some restoration techniques and practices work, and, equally important, why some fail (Thayer et al. 2003, Saldi-Caromile et al. 2004). Monitoring can be a powerful tool if the objectives are clearly stated and the monitoring action is well-designed, rigorous and scientifically-based (van Diggelen et al. 2001). Monitoring must be scaled spatially and temporally to the response variables assessed (White and Walker 1997, Block et al. 2001). Additionally, to provide unbiased estimates of significant response variables, the monitoring design needs to include statistical considerations, such as the distribution of sampling sites and the number of replicate samples to be collected (Gibbs et al. 1999).

Although Mediterranean countries have long-standing experience in reforesting degraded lands, assessment and monitoring have rarely been done, and when they were done, they were essentially based on a few early assessments of seedling survival. Consequently, scientific and technical information on past reforestation projects is scarce (see Chapter 4, this volume). Some recognised reasons for this limited monitoring of forest restoration actions in the Mediterranean region include economic and political constraints as well as scientific and technical limitations. Thus, for example, there are many tree species for which our knowledge is still extremely limited (Méndez et al. 2008), and there is still a lack of understanding of the effects of reforestation on various system processes and components (Maestre and Cortina 2004), which compromises the selection of response variables and critical values for monitoring. Monitoring of forest restoration should be carried out over a a long enough time to address the processes and dynamics of interest, and to take into account the variation in environmental conditions (Block et al. 2001). Unfortunately, there are imposed constraints associated with political turnovers and the time frames for agency budgets. Consequently, monitoring efforts are not supported with the funding required to implement long-term monitoring plans. Despite the recognised constraints, monitoring needs to be a routine practice and an integrated component of restoration planning. Effective monitoring results in greater efficiency and lower cost for future restoration activities, provides technical bases to demonstrate the effectiveness of public expenditures, and informs funding agencies so they can refine their funding priorities over time (Gaboury and Wong 1999).

Though the uncertainty in forest restoration can probably never be fully overcome (Gomez and Elena 1996, Pemán 1997, Navarro et al. 2003b), periodic estimates of the magnitude and trajectory of suitable response variables provide an ongoing evaluation of the restoration strategy and, thus, a basis for decisionmaking under an adaptive management framework. The restoration activity can thus be improved, creating a feedback loop of continuous learning (Gibbs et al. 1999, Gayton 2001), where the problems addressed, the objectives, the design and implementation of the project and the monitoring program are adjusted to reflect the understanding gained through the monitoring action (Fig. 1). Actually, the monitoring action should permeate the whole restoration process, from problem assessment and project design to evaluation and adjustment. Thus, for example, to assess initial conditions for the target area, pre-restoration (baseline) monitoring must occur. Monitoring procedures for evaluating project implementation and project results must also be established. Monitoring includes decision making. Thus, monitoring programs should incorporate all the procedures that connect the monitoring results to the decision process (Noon et al. 1999), and decisionmakers should be involved in the planning and application of the monitoring programs (Noble and Norton 1991).

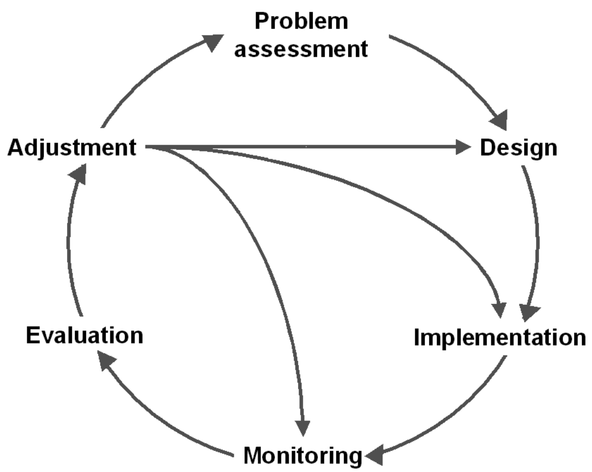

FIGURE 1. An adaptive management framework for ecosystem restoration (adapted from Gayton 2001).

Monitoring phases: baseline, implementation, effectiveness, and long-term monitoring

The herein-proposed monitoring methodology to be applied to forest restoration in the Mediterranean region follows the general monitoring approach suggested by Holl and Cairns (2002), and can be described as an adaptive management-based method articulated in four key phases: Baseline, Implementation, Effectiveness, and Long-term monitoring. Baseline monitoring primarily addresses the assessment of baseline, pre-restoration conditions. Parallel to the planning of the restoration project, monitoring objectives and questions, and a strategy to answer these questions, are also established during this first phase of baseline monitoring. Frequency and duration of monitoring measurements, spatial scales for monitoring, and budget opportunities and constraints are key elements to be considered while designing the monitoring plan. Implementation monitoring is used to assess whether the previously established quality control conditions for the project implementation are being met. This phase serves as a checklist for implementing and managing the restoration project, and for assuring compliance with the contract prescriptions. Effectiveness monitoring assesses the effect of the restoration action on target attributes previously selected as suitable indicators for evaluation purposes. This phase helps in determining the degree to which restoration activities attain the specific objectives set out in the planning phase. Finally, long-term monitoring aims at assessing trajectories and trends in the restored area, determining whether the ultimate restoration goals can be attained or whether the key assumptions underlying the restoration project were valid. Table 1 describes some primary tasks for each of these phases. Each task can also be considered a contingency factor that influences the quality of the implementation of the next steps in the monitoring program.

The application of this four-phase monitoring approach to forest restoration projects in the Mediterranean region can be further improved by using simplified but technology-sound based tools (e.g., using PERT® charts to schedule project activities – see Fig. 2; using GPS technology and GIS applications to consider and analyse spatial information), and by establishing monitoring objectives according to the existing empirical experience in Mediterranean forest restoration, and the particular environmental and socio-economic framework in the region. The following sections describe implementation examples of the various monitoring phases in the context of forest restoration in the Mediterranean region.

TABLE 1. Examples of primary tasks associated with key phases in forest restoration monitoring.

Monitoring phases	Tasks
Baseline monitoring	- Before the start of the restoration project, identify existing biophysical, social and cultural conditions and establish benchmarks (gather information on historical conditions and current land uses, identify and analyse available cartography, aerial photography, etc.) - Document project decision-making process (review available reports on site conditions and management plans) - Document project objectives and design (including time and spatial scales considered) - Become familiar with similar restoration projects - Define monitoring questions and develop a monitoring program
Implementation monitoring	- Assess implementation technique (e.g., seedling quality, site preparation, treatment application, spatial arrangement and scheduling of treatment application, etc.) - Assess compliance with the contract
Effectiveness monitoring	- Choose a standardised sampling design and monitoring variables that are based on a conceptual model for ecosystem response to restoration - Adjust monitoring design to the temporal and spatial scales of the processes addressed. Define the target sampling population and plot size and shape (consider potential for establishing a pilot study in the area), and the statistical parameters to be considered - Evaluate project results according to benchmarks and specific project objectives - Analyse, interpret, and summarise results; deliver monitoring reports and provide feedbacks for the adjustment of restoration objectives, design, and monitoring plan.
Long-Term monitoring	- Select suitable response variables for long-term monitoring - Continue the monitoring action in the long-term; establish permanent plots and sampling points. - Evaluate the ultimate goal, strategies, and cost-effectiveness of the restoration effort - Consider and propose alternative approaches in view of the lessons learned from the monitoring action

Baseline monitoring

Baseline monitoring creates an image of the existing conditions before the restoration work begins. It should include two critical steps: gathering of available baseline data and data validation. Baseline data validation aims at verifying in the field the information provided by available cartography and reports, and thus establishing the actual site conditions. Both the collection of baseline information and the validation take place before the implementation-monitoring phase. The main objectives of baseline monitoring are (1) to establish initial conditions and benchmarks to be considered in the subsequent monitoring phases; (2) in combination with implementation monitoring, to detect deviations between what has been planned and what is actually being implemented in the field; and (3) to provide the necessary background information to design the appropriate monitoring program.

An efficient software that can greatly facilitate monitoring planning efforts is Microsoft Project®, with added assistance from its Program Evaluation Research Technique (PERT) extension tool. This tool helps to plan all the tasks that must be completed as part of a program, to determine a realistic duration for them, and to monitor the achievement of project goals. A PERT analysis can also be used to compare the proposed planning of the restoration project with what actually takes place in the field, and thus to detect deviations during project implementation. The importance of these deviations is well-illustrated with the example of PERT charts created for an actual post-fire restoration project in Andalusia (Fig. 2). The application of PERT® also allows us to foresee any potential error regarding project planning and scheduling (timing, order of implementation), which can thus be corrected while the project is being implemented.

Limitations to baseline monitoring include difficulties in gathering data, particularly in digital format (e.g., readily available spatial information such as digital cartography); moreover, budgeting constraints often reduce the necessary field work involved to achieve the desired project analysis. The latter case applies to all monitoring phases.

Finally, consideration must be given to the following fundamental questions regarding monitoring design. These questions must be answered before starting any restoration work in order to progress toward the next monitoring phases:

1. Are the monitoring objectives and questions clearly established?
2. Is there a suitable monitoring strategy developed to meet those objectives?
3. Do the proposed activities and trials properly address the questions that the monitoring is supposed to answer?

Implementation monitoring

Implementation monitoring aims both to assess whether what is being implemented is correct or not (according to the project prescriptions) and to evaluate the potential impact of the deviations observed in the degree of achievement of the restoration project goals. For common planting-based forest restoration projects, actions to be revised at the implementation monitoring stage include the selection and use of the plant material (i.e., species selection, seed provenance, plant quality, stock handling and transportation); treatment application (i.e., selection of planting sites and microsites concerning aspect, slope, soil properties, rock outcrops; field site preparation; post-planting treatments, etc.); and actual scheduling of the implemented actions.

Project implementation activities are critical to the overall success of the restoration project, and therefore they deserve careful assessment. It is often stated that immediate failure after outplanting is the result of uncontrollable conditions (e.g., climate anomalies, uncontrolled grazing, floods, etc.) affecting the area to be restored. However, the lack of

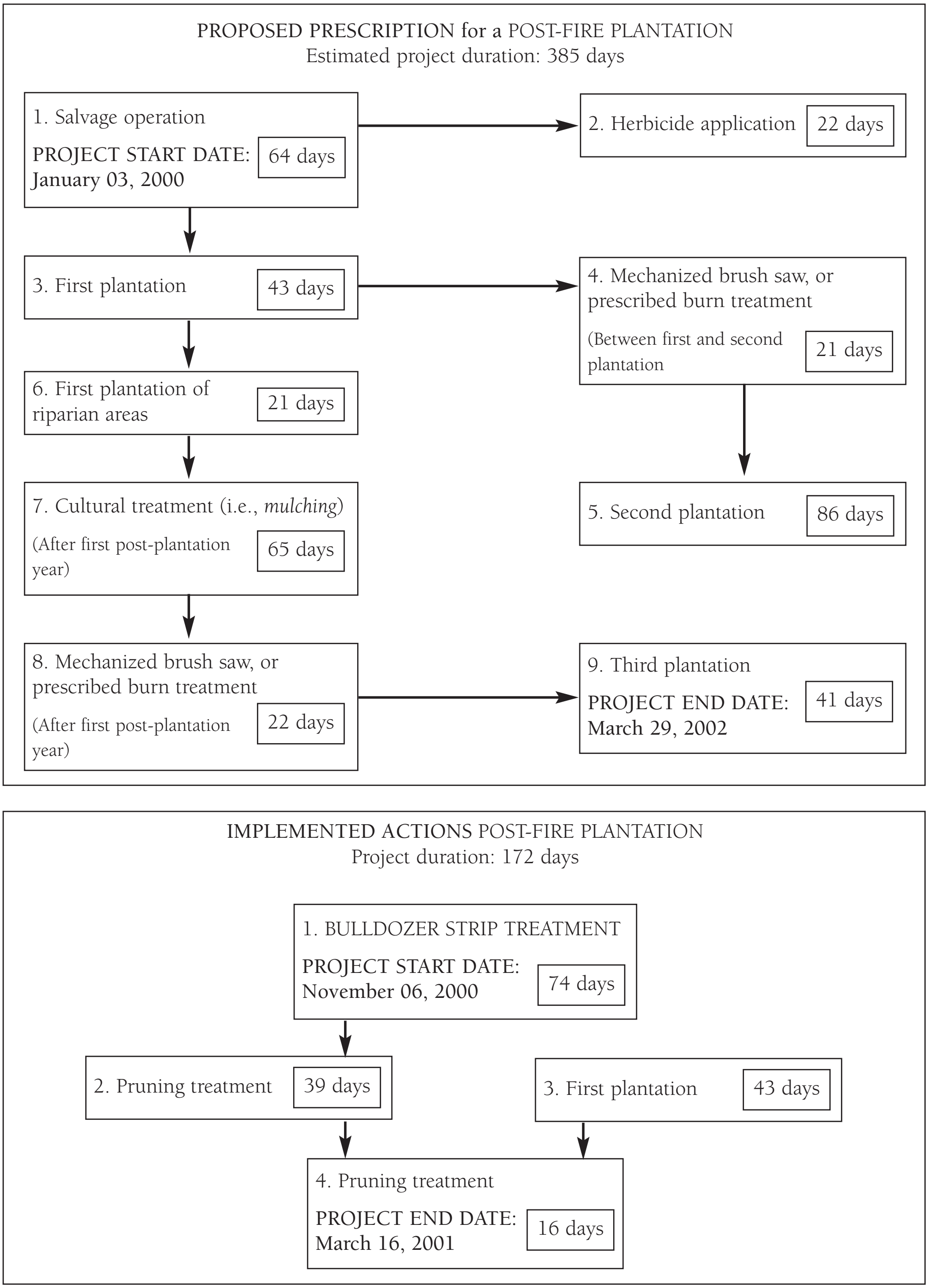

FIGURE 2. PERT chart of proposed prescription versus implemented actions for a post-fire reforestation project in El Madroñalejo (Seville, southern Spain).

implementation monitoring may leave the real causes of post-planting seedling mortality indeterminate. For example, poor quality nursery stock (e.g., unbalanced shoot to root ratios) that jeopardizes plant survival under field conditions could be identified by appropriate monitoring during the implementation phase of the project. Table 2 shows some results from the assessment of short-term post-plantation seedling survival for the various nursery stock types used in a post-fire restoration project. The assessment results help to identify species and seedling stock combinations that might not be suitable for the conditions prevailing in the area to be restored.

Although quality standards for seedling nursery stocks do exist for many of the Mediterranean species used in common restoration projects, they are frequently ignored. As a result, the quality of the seedling stocks produced for planting-based projects is often very poor. Implementation monitoring checks whether these seedling quality standards have been met or not. Also, regardless of the degree of compliance with the standards achieved, it comparatively assesses the effect of the various stocks used on early seedling performance (Table 2).

TABLE 2. Average survival rate (percentage) for contrasting nursery stocks (various combinations of species and container sizes) used in a post-fire restoration project in Guadiamar, south-western Spain (Navarro et al. 2003a). Data obtained 3 months after outplanting.

	Volume of seedling container				
Species	**200 cc**	**210 cc**	**300 cc**	**400 cc**	**500 cc**
Arbutus unedo		0 %		5 %	
Crataegus monogyna			40 %		
Pyrus bourgaeana			15 %		
Pistacia lentiscus		95 %	90 %		35 %
Retama sphaerocarpa	95 %	100 %		100 %	

Temporal and spatial scales for project monitoring are key aspects to be considered when scheduling the monitoring program, and the implementation monitoring in particular. Both the monitoring design and the size, number and shape of the monitoring plots must be scaled to the extent of the target area, and the questions being addressed (Holl and Cairns 2002). Depending on the size of the project and the degree of regularity of the activities performed, the frequency of monitoring (when and how often monitoring activities will occur during project implementation) can vary from intensive quantitative data collection during certain periods of the implementation phase to more infrequent and regularly scheduled measurements (Gomez and Elena 1996, Pemán 1997, Navarro et al. 2003b).

Although the implementation phase of a restoration project is critical to the achievement of the project goals, deviations from project design and implementation schedule are very common (Fig. 2). Reasons that account for such deviations include: (a) lack of continuity between planning and implementation, since the manager implementing the project in the field is not usually the designer of the project; (b) lack of adequate baseline data needed in advance to design the restoration project to properly match the site conditions; and (c) budgetary constraints that preclude making needed adjustments to address unforeseen problems that emerge during the implementation phase. For example, Figure 3 illustrates the within-site spatial variation in the degree of implementation (relative to the planned actions) for a particular post-fire restoration project, with a spatial arrangement of treatment units that ended up differing considerably from the planned units due to the challenges of field implementation.

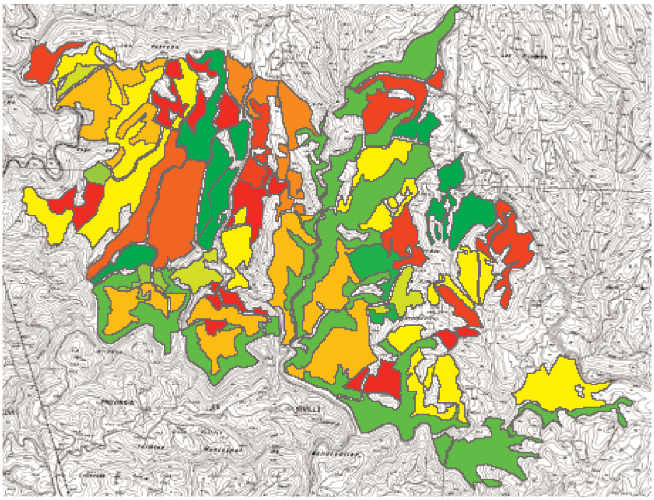

FIGURE 3. Differences between planned and implemented restoration actions and project stages in the El Madroñalejo post-fire restoration project, Seville, southern Spain. Different colours represent different degrees of implementation success.

Feedbacks from the implementation monitoring to managers must serve to determine if the modifications produced are significant and relevant and whether they should be reversed or altered. The modifications accepted, and the respective changes in the associated specific objectives, will result in a new (modified) framework for designing and implementing the subsequent monitoring phases.

Effectiveness monitoring

During the effectiveness monitoring phase, changes and trends over time in one or more of the selected indicators are assessed and evaluated (White and Walker 1997). Effectiveness monitoring is used to determine whether the restoration project achieved the specific project objectives. In reforestation and afforestation projects in the Mediterranean region, effectiveness monitoring is typically performed during the first few years after the implementation phase of the project and, therefore, is focused on the initial response of the target area and species, assuming that this short-term response is a good indicator of the long-term trends. Effectiveness monitoring requires data collection to follow a repeated sampling approach, in order to provide preliminary information regarding the potential dynamics in the target area. This is particularly applicable in the context of adaptive management, where the implementation and management of the restoration efforts depend on monitoring feedback. Furthermore, established monitoring plots may act as study sites that provide observational data for future consideration.

Field measurements of seedling survival and growth are the first steps in evaluating the degree of success of most planting-based restoration projects in the Mediterranean region (Maestre and Cortina 2004). These measurements aim to evaluate the degree of seedling establishment as well as the seedling response to the conditions prevailing during the first post-planting years, with special emphasis on the critical period corresponding to the first summer season. Survival and growth values are then analysed in relation to meteorological and site conditions, site preparation, microsite location, treatments applied, and data available from the previous monitoring phases. These analyses should provide insights into cause-and-effect relations between environmental stressors, treatments applied and seedling response (Machmer and Steeger 2002). For example, seedling survival was the key success indicator for the Guadiamar post-fire restoration project (Table 3). By comparing this project with similar restoration projects and previous experiences in similar areas, and taking into account the weather conditions during the first post-planting year, we used these survival values to estimate future project outcomes regarding ecosystem functioning and plant population trends.

As in to the implementation-monitoring phase, the schedule of the effectiveness-monitoring plan can be adjusted to allow more intensive, quantitative data collection to take place during pre-determined critical periods, such as the first post-summer season. After this initial period, the frequency of monitoring can be reduced to address long-term dynamics and silvicultural needs rather than initial success (see below).

Preparation of data summaries and interpretive reports, and associated feedbacks to management are also key components of effectiveness monitoring (Gaboury and Wong 1999, Mulder et al. 1999). Appropriately analysed information should be rapidly accessible to a wide audience, particularly to decision makers. Data summaries should be brief, comprehensive reports on the essential data collected; periodic interpretive reports should evaluate the significance of the status and trends emerging in the monitoring data. The resulting information can thus be used to change plans and directions, as well as budgetary decisions.

TABLE 3. Survival rate (percentage) for the target species used in the Guadiamar post-fire restoration project in south western Spain (Navarro et al. 2003b).

	Number of seedlings		Survival (%)
Species	Estimated (to be planted)	Actually planted (2002)	1 year after planting (2003)
Arbutus unedo L.	1650	985	46
Celtis australis L.	1672	797	44
Ceratonia siliqua L.	494	256	41
Chamaerops humilis L.	144	41	21
Cistus salviifolius L.	550	389	64
Cotoneaster integerrimus Medicus	1700	420	19
Crataegus monogyna Jacq.	1150	563	46
Fraxinus sp.	1902	918	23
Genista sp.	888	378	36
Lavandula angustifolia Miller	2550	1245	38
Lonicera sp.	300	55	14
Myrtus communis L.	2050	1259	56
Nerium oleander L.	200	161	79
Olea europea L.	2435	1491	60
Pinus sp.	575	383	47
Pistacia lentiscus L.	814	456	48
Populus sp.	478	361	59
Pyrus bourgaeana Decae	1200	736	51
Quercus ilex L.	460	281	56
Rosa sp.	1100	577	49
Rosmarinus officinalis L.	550	422	74
Rubus fructicosa L.	600	95	2
Salix atrocinerea Brot.	250	227	80
Tamarix sp.	150	115	69
Teucrium fruticans L.	888	476	43
TOTAL	**24428**	**11305**	**47**

Long-term monitoring

Long-term monitoring may be necessary to evaluate the results of many restoration projects. In fact, the final outcome of a reforestation project can only be assessed comprehensively in the long term, after several decades. However, funding sources for long-term monitoring are very limited. Therefore, for the majority of forest restoration projects in the Mediterranean region, monitoring programs rarely last more than four-five years, despite many of these projects consider long-term projections (e.g., 50–100 years for common forest restoration projects based on plantations of tree species).

Long-term monitoring builds upon the previous effectiveness-monitoring phase by increasing the number of parameters assessed and the temporal scale of the measurements. Monitoring plots installed during the first stages of a restoration project can be used to establish a permanent monitoring structure. The purpose of longer-term monitoring in such established plots is to assess the recovery trajectory and self-maintenance of the target area after the implementation of the restoration project, and to evaluate the effect of the silvicultural treatments applied in the area. The feedbacks provided by long-term monitoring are of particular interest for adjusting post-project management practices aimed at modulating the succession processes to ensure that long-term conservation and restoration goals are met (Harrington 1999).

In addition to the common variables used for the short-term effectiveness monitoring, such as seedling survival and growth in planting-based projects, a wide range of structural and functional indicators are appropriate for long-term monitoring (see Chapter 4, this volume). For example, plant cover, density and biomass, diversity of plants and fauna (including wildlife abundance and diversity), and indicators of ecosystem processes, such as biological interactions (e.g., pollination, dispersal), fire incidence, forest pests, and soil organic matter, have often been used to evaluate restoration success (Ruiz-Jaén and Aide 2005).

Conclusions

The growing number of restoration projects worldwide has increased the interest in monitoring and evaluating of these efforts. However, monitoring still seems to be carried out in only a small proportion of restoration projects, only a small number of projects require or mandate performing project monitoring, and in most cases the monitoring effort only addresses compliance with the contract prescriptions. Environmental and economic constraints on forest restoration make it necessary to place more value on project monitoring. The monitoring approach presented in this chapter can easily be applied to common forest restoration projects in the Mediterranean region in the context of an adaptive management framework. If monitoring is well-designed and conducted, it helps to reduce uncertainties, assisting and improving the restoration practice. Project sponsors, funding agencies, and managers must ensure the implementation of appropriate monitoring programs, which may be critical to future progress and funding of forest restoration in the region.

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III.

Innovative approaches in forest restoration

Genetic Quality of Forest Reproductive Materials in Land Restoration Programmes

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Introduction

The choice of plant reproductive materials for restoration programmes include seeds and fruits, whole plants and parts of plants to be used as cuttings for vegetative propagation. The choices made are – or should be – based not only on external aspects but also on genetic characteristics. External quality could be easily evaluated by the user, but genetic quality depends on factors such as genetic diversity, selection criteria for different traits, and a number of biophysical components where the material was collected, none of which are directly observable. The user, therefore, has to rely on the information provided by the collector, supplier, marketing company, or some control authority in charge of the application of the regulation on marketing of the plant reproductive material. The EU scheme on marketing of forest reproductive material is applied to 47 species (or genera) in

TABLE 1. Species under regulation by the Directive 1999/105/CE on marketing of forest reproductive material.

Abies alba Mill.	*Larix decidua* Mill.	*Prunus avium* L.
Abies cephalonica Loud.	*Larix x eurolepis* Henry	*Populus* spp.
Abies grandis Lindl.	*Larix kaempferi* Carr.	*Pseudotsuga menziesii* Franco
Abies pinsapo Boiss.	*Larix sibirica* Ledeb.	*Quercus cerris* L.
Acer platanoides L.	*Picea abies* Karst.	*Quercus ilex* L.
Acer pseudoplatanus L.	*Picea sitchensis* Carr.	*Quercus petraea* Liebl.
Alnus glutinosa Gaertn.	*Pinus brutia* Ten.	*Quercus pubescens* Willd.
Alnus incana Moench.	*Pinus canariensis* C.Smith	*Quercus robur* L.
Betula pendula Roth	*Pinus cembra* L.	*Quercus rubra* L.
Betula pubescens Ehrh.	*Pinus contorta* Loud.	*Quercus suber* L.
Carpinus betulus L.	*Pinus halepensis* Mill.	*Robinia pseudoacacia* L.
Castanea sativa Mill.	*Pinus leucodermis* Antoine	*Tilia cordata* Mill.
Cedrus atlantica Carr.	*Pinus nigra* Arnold	*Tilia platyphyllos* Scop.
Cedrus libani A.Richard	*Pinus pinaster* Ait.	
Fagus sylvatica L.	*Pinus pinea* L.	
Fraxinus angustifolia Vahl.	*Pinus radiata* D. Don	
Fraxinus excelsior L.	*Pinus sylvestris* L.	

all the European countries (Table 1) when used for forestry purposes, but each country can regulate additional species in their territory (e.g., Spain has added 20 Mediterranean species). There are many other species, which could not be under regulation, but the principles of this scheme are valid and can be considered when selecting propagation material for restoration programmes. Therefore, in those cases, it would be desirable to precisely define the type and characteristics of the basic material from which the reproductive material should be collected.

The basic principles on which the regulation is based can be summarised as follows:

a) The existence of diversity at different levels (species, populations, and individuals). The genetic diversity among species is easily recognised, but differences among populations are in some cases neglected even they are large for many important traits (Fig. 1). Langlet (1971) presented an historical overview on the differentiation among populations in forest trees, and this genetic variation can be influenced by different life-history traits of the species under consideration (Hamrick 1992). The genetic differences among individuals are easily recognised in many forest species, and especially in those of commercial interest with different breeding programmes (***Populus***, ***Salix***, ***Castanea***, ***Juglans***, ***Pinus***, ***Picea***, among others).

b) The importance of some characteristics of the basic material in the future performance of the plantations, specially the origin, the diversity and the selection processes to which populations have been submitted.

c) The difficulty in assessing such characteristics, and the necessity of an efficient control system at the European level. This control system covers the entire production chain, from seeds to plants, in order to avoid fraud in the commercialization process.

Provenance: ***Coca***, Spain ***Tamjout***, Morocco ***Leiria***, Portugal

FIGURE 1. Differences among three maritime pine (***Pinus pinaster*** Aiton) populations under common garden experiments (in a provenance test conducted in Cabañeros, Ciudad Real).

The genetic quality of the reproductive material should, therefore, ideally be based on sound knowledge of the genetic basis of the processes of selection and characterization of the plant materials under consideration, as well as by evaluation of the material under common garden conditions. This of course is not always possible.

Genetic basis of breeding

The use of species with advanced breeding programmes (i.e., those with several breeding generations) in restoration is usually of limited importance, but we have to understand the principles of selection for the choice of the best reproductive material.

The genetic basis of breeding has been described in different papers (e.g., Zobel and Talbert 1988, Alía et al. 2005). The breeders depend on the existence of phenotypic variability among the individuals, the degree of genetic control (heritability) of the traits of interest, the selection (based on phenotypic or genetic evaluation) of some individuals from the population with desirable properties, and possible crosses among those individuals to advance in breeding (Fig. 2).

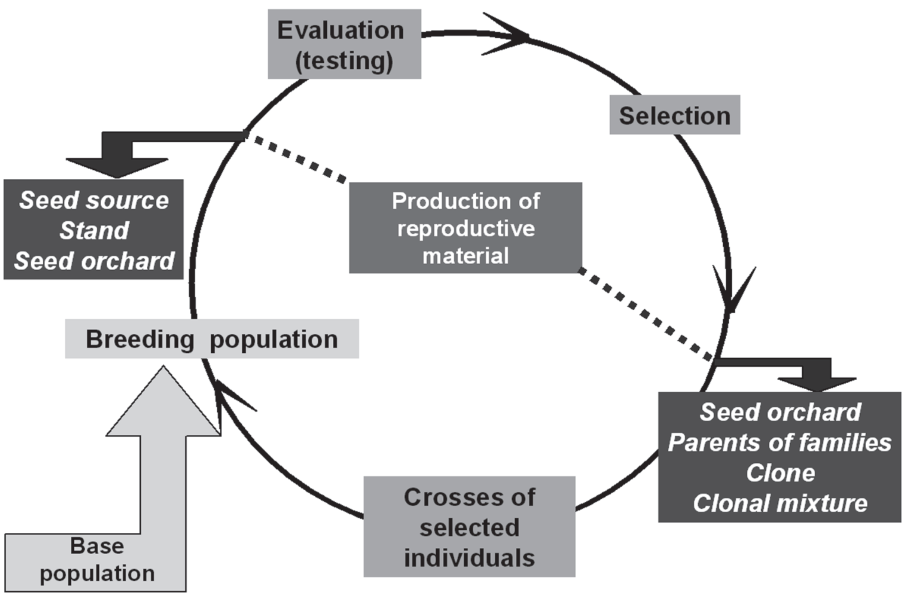

FIGURE 2. Different stages in a breeding cycle.

To be effective, breeding has to be focused on traits of importance, with high heritability (Table 2) and low correlation with undesirable traits. For tree taxa of high economic or social importance, e.g., ***Populus***, ***Juglans***, ***Castanea***, ***Prunus***, ***Ulmus***, ***Pinus***, ***Picea***), the most important traits are those related to growth, drought or frost tolerance, and wood properties.

Origin and Regions of provenance

One of the central concepts for the use of forest reproductive material is the ***origin*** of the material (see Box 1 for definitions, and Table 3). The origin determines many important characteristics related to the future performance of the plants (e.g., traits related to adaptation to climate, traits related to adaptation to biotic or non biotic factors, growth, survival), as a result of the evolutionary factors that shape the genetic structure of the populations in the forest species. Many studies have demonstrated a high level of

TABLE 2. Heritability values for different traits (modified from Alía et al. 2005).

Trait	**Species**	**Heritability**
Drought tolerance	*Pinus pinaster*	High
	Castanea sativa	High
Phenology	*P. x euramericana*	High
	Populus alba	Moderate
	Populus deltoides	High
	Populus deltoides	Moderate
	Castanea sativa	High
Height	*Pinus sylvestris*	Low
	Pinus halepensis	Moderate
	Pinus pinaster	Low
	Castanea sativa	Moderate
	P. x euramericana	Moderate
Diameter	*Pinus sylvestris*	Low
	Pinus nigra	Low
	Pinus pinaster	Low
Wood density	*Pinus sylvestris*	High
	Pinus pinaster	High
	P. x euramericana	High
Form	*P. x euramericana*	High
	Populus alba	Very Low
	Pinus pinaster	Moderate
Branching	*P. x euramericana*	High
	Populus alba	Moderate

Very Low: 0-0.1; Low: 0.1-0.35; Moderate: 0.35-0.6; High: 0.6-0-9; Very High: 0.9-1.0

differentiation among populations for traits under selection (such as bud set, growth initiation and cessation, frost tolerance, and drought tolerance) (e.g., Van Andel, 1998). However, the characters related to migration, isolation, genetic drift (i.e. not under selection) present a variable level of differentiation.

Box 1. Definitions (Source: Directive 199/105/CE on marketing of forest reproductive material):

(a) ***Autochthonous stand or seed source***: An autochthonous stand or seed source is one which has been continuously regenerated by natural regeneration. The stand or seed source may be regenerated artificially from reproductive material collected in the same stand or seed source or autochthonous stands or seed sources within the close proximity;

(b) ***Indigenous stand or seed source***: An indigenous stand or seed source is an autochthonous stand or seed source, or a stand or seed source raised artificially from seed, the origin of which is situated in the same region of provenance.

(c) ***Origin***: For an autochthonous stand or seed source, the origin is the place in which the trees are growing. For a non-autochthonous stand or seed source, the origin is the place from which the seed or plants were originally introduced. The origin of a stand or seed source may be unknown.

(d) ***Provenance***: The place in which any stand of trees is growing.

(e) ***Region of Provenance***: For a species or sub-species, the region of provenance is the area or group of areas subject to sufficiently uniform ecological conditions in which stands or seed sources showing similar phenotypic or genetic characters are found, taking into account altitudinal boundaries.

TABLE 3. Origin and provenance of the basic material, and the forest reproductive material obtained.

Origin	**Provenance**	
Basic Material	**Basic Material**	**Reproductive material**
Site A	Site A	Autochthonous (origin = provenance)
Site B	Site A	Non Autochthonous (origin ≠ provenance) Known origin (origin =B)
?	Site A	Non Autochthonous (origin ≠ provenance) Unknown origin

It is necessary to delineate regions of provenance for each species, i.e. zones with similar ecological characteristics (Box 1). These regions are the base of the marketing for source-identified and selected forest reproductive materials (see description of the categories below). Two methods (agglomerative and divisive) have been followed to establish the regions of provenance in Europe (Fig. 3), and they are available for the different European countries. The description is available in different monographs or webpages from the different designated authorities in each country (e.g., CEMAGREF 2003 for France; Martín et al. 1998, García del Barrio et al. 2001, 2004 for Spain).

a) ***Divisive method***: the territory is divided into disjoint ecologically homogeneous regions, taking into account climatic, geographical, and soil traits related to the performance of the species under consideration. This method has been applied in different European countries (see Gordon 1992, CEMAGREF 2003, García del Barrio et al. 2001, 2004). This method has the main advantage of defining the same regions for all the species under consideration, but it does not take into account some possible special characteristics of the species (e.g., patterns of genetic variation, distribution patterns).
b) Agglomerative method. The stands of a species with similar phenotypic, genetic or ecological characteristics are grouped to form a region of provenance. Therefore, each species has different regions of provenance, but they describe more precisely the pattern on known variation of the species. This method can be used for species with precise information on phenotypic, genetic or ecological variation.

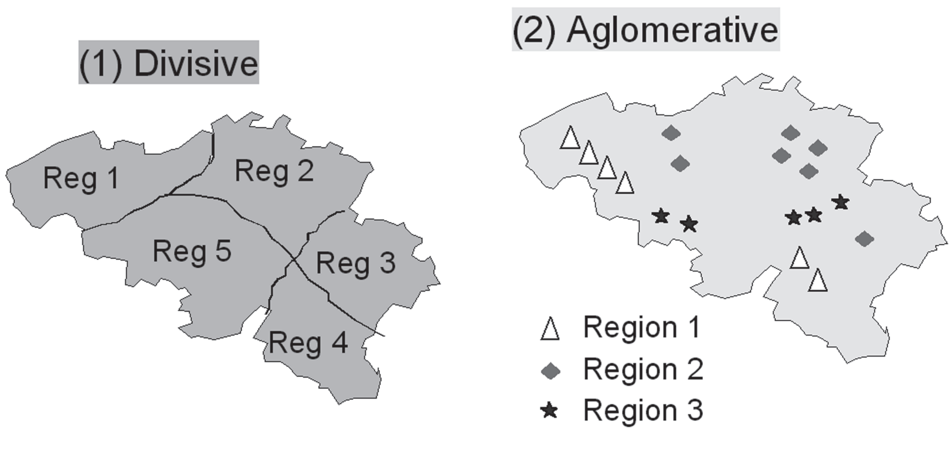

FIGURE 3. Methods applied for the delineation of regions of provenance (CEMAGREF 2003, Alía et al. 2005).

The main characteristics of a system of regions of provenance are the following:

– ***The region of provenance determines the geographical limits from which reproductive material can be mixed for commercialization.*** Both source-identified and selected reproductive materials have to be collected in seed sources or stands from one region of provenance. They can not be mixed with material from other regions.
– ***The region of provenance simplify the marketing of forest reproductive material by identifying zones in which seed or fruits have been collected.*** For a species as ***Pinus sylvestris*** there are 17 regions of provenance in Spain, and more than 400 seed sources or stands. Therefore, it is easier for the user to recognise the 17 regions of provenance instead each of the seed sources or stands.
– ***The region of provenance simplifies the seed transfer rules in national forestation programmes***. Usually, seed transfer rules are based on information on a limited number of materials evaluated in a limited set of ecological conditions. Therefore, it is much easier to define the rules for each region instead of rules for specific seed sources or stands.
– ***The region of provenance can be used for planning breeding or conservation activities***. A region, or several regions, can be combined to consitute a breeding zone in a breeding or conservation programme.
– ***The practical importance for marketing is different for each region, and therefore they are not used similarly in restoration programmes.*** Some regions are susceptible to be used in a broad spectrum of ecological situations, whereas some others are restricted to a local used.

Taking into considerations those different aspects, regions of provenance are the first step in the selection of the material for improvement and restoration programmes, and a deep knowledge on the ecological, phenotypic or genetic characteristics of the different regions is essential for a correct choice of the material to be used in each case. After identifying the most suitable region of provenance we have to decide on other characteristics of the material as type of basic material and on the category of the forest reproductive material.

Characteristics of basic and forest reproductive materials

Forest reproductive materials have to be collected in specific types of basic material (if the species is under regulation), in order to guarantee some genetic properties, and those basic materials have to be included in the National Registers of approved basic material for the production of forest reproductive material. The national registers are available from the different national authorities.

The EU certification scheme distinguishes six types of basic materials: seed sources, stands, seed orchards, parents of families, clones and clonal mixture (see Box 2), This is similar to other control schemes for the international trade as OECD (Organisation for Economic Co-operation and Development) and AOSCA (Association of Seed Certifying Agencies).

Box 2. Types of basic materials. (Source: Directive 1999/105/CE on marketing of forest reproductive material, adapted from Nanson 2001).

(a) ***Seed source***: Seed is collected within a zone of collection called a seed source. This zone is not necessarily delineated, nor clearly identified. On the contrary, the Region of Provenance where it lies has to be clearly delineated and identified in a National Register (maps).

(b) ***Stand***: It is a well delineated population of trees possessing sufficient uniformity, and referenced in a National Register.

(c) ***Seed orchard***: It is a plantation of selected clones or families which is isolated and managed to avoid or reduce pollination from outside sources, and managed to produce frequent, abundant and easily harvested crops of seed. There are two main types of Seed Orchards: i) Clonal Seed Orchards, ii) Family Seedling Seed Orchards. These last are in fact progeny tests with small plots, the trees of which are later submitted to genetic selective thinning.

(d) ***Parents of families***: They are defined groups of trees (clones) producing open pollinated or controlled pollinated families. These families are afterwards mixed for production. Most often, this mixture of families is vegetatively bulk propagated (e.g.: cuttings of *Picea sitkensis* in Great Britain).

(e) ***Clone***: It is a group of individuals (*ramets*) derived originally from a same single individual (*ortet*) by vegetative propagation, for example by cuttings or micropropagation. Individuals of the same clone have the same genotype, unless somatic mutation or error.

(f) ***Clonal Mixture***: It is a mixture in defined proportions of initially identified clones. Usually, the ramets of these clones are mixed, bulked and so delivered for afforestation. The clonal identity of the individual ramets is therefore generally lost at the forest stage and often already at the vegetative propagation stage. In current scientific language, clonal mixtures are usually denominated as "multiclonal varieties" or "polyclonal varieties".

The four categories of forest reproductive material depend on the selection method and the evaluation processes of the basic material (Nanson 2001, Alía et al. 2005). They have to be labelled with different colour when commercialized.

– *Source Identified (yellow label)*. The reproductive material is derived from seed sources or stands from one single region of provenance. Basic materials have thus not been submitted to any selection and are only identified by the region of provenance.

– *Selected (green label)*. The basic material has undergone a phenotypic selection at the population level. It is the case of selected stands that are phenotypically superior to stands of the same region of provenance. Presently, the basic material related to this category is still representing the major part in the world, often more than 90% of basic materials in national catalogues.
– *Qualified (pink label)*. Components (trees, clones) of relevant basic materials must have undergone a phenotypic selection at the individual level. Seed orchards are the most common material in this category.
– *Tested (blue label)*. Forest Reproductive Materials produced by relevant basic materials must be found genetically superior, by comparative testing or by an estimate of the superiority of the reproductive material calculated from the genetic evaluation of the components of the basic material.

The EU certification scheme defines the types of basic material accepted for producing forest reproductive material from the different categories (Table 4). Those different reproductive materials differ in terms of genetic diversity (higher for seed sources or stands, lower for clones), genetic gain (higher for clones of clonal mixtures, lower for seed sources), and for the degree of phenotypic and genetic evaluation of the materials (higher for tested material, lower for identified materials). However, the certification scheme does not mean a scale, in which the "value" of the reproductive materials increases from source-identified to tested materials. The source-identified and selected materials are more useful than tested materials for many restoration programmes as they have a determined origin, no or slight selection and high levels of genetic diversity.

TABLE 4. Categories under which reproductive material from the different types of basic material may be marketed (EU Directive 199/105/CE).

Type of basic material	**Category of the forest reproductive material**			
	Source Identified	Selected	Qualified	Tested
Seed source				
Stand				
Seed orchard				
Parents of families				
Clone				
Clonal Mixture				

Use (*transfer rules*) of forest reproductive material

When deciding the best reproductive material to use in a given restoration programme it is necessary to choose among a list of materials (seeds or plants) available from different providers. These will differ in their basic offerings in term material, region of provenance and categories. It is necessary to take into consideration different factors:

- *Which is the deployment zone?* The main ecological characteristics of the area where the material will be established determine some of the properties of the material to use.
- *Which is the best procurement seed zone (region of provenance)?* To select the origin of the forest reproductive material (for source-identified and selected reproductive materials) it is possible to use information on the ecological, phenotypic and genetic information of the basic material from the different regions of provenance. However, for qualified or tested material, the origin is not so important, as the material can be the result of advanced selection programmes where the pedigree of the material is much more important to decide on its properties. Some additional principles, mainly related to the conservation of forest genetic resources, have to be considered in order to avoid endangering valuable local resources by introducing exotic material. Even when the regulation on forest reproductive material is not aimed at conservation purposes, this aspect should be considered as a general principle in all restoration programmes.
- *Do we need an increment in the mean value of some important traits? (i.e., an increment in wood production or survival)*. In commercial plantations, in general, the goal is to maximize production for some characteristics. Therefore, qualified or tested materials are preferred because they have been selected for those characteristics and have been tested under ecological conditions similar to those of the plantation sites. However, when the objectives are related to ecological restoration, other features such as adaptation to some special conditions, or the use of specifically local adapted material may be the top priority. In absence of such materials, selected materials could be more suitable in such conditions.
- *Which level of genetic diversity do we need?* The genetic diversity of the material to be used is important, especially if we are interested in plantations at the long term, and where the natural regeneration can have an important role in the future. The different types of basic material can yield material with different levels of diversity, i.e., with low or very low levels for parents of families, clones or clonal mixtures.

From all these factors, we shall now focus on the relationship among deployment and procurement zones (Buijtenen 1992), as the other factors depend mainly on the specific objectives of the restoration programme. Deployment and procurement zones (region of provenance) can be the same (if using local sources) or different (if using exotic material because of some interesting traits or absence of a desirable local source).

To establish the transfer rules, there are some general patterns established as a result of many transfer experiments (see Zobel and Talbert 1988, as an example):

– Local seed sources are usually locally adapted (Kawecki and Ebert 2004).
– Local seed sources are not usually the most productive (Namkoong 1969).
– We have to take into considerations the seasonality in rainfall and temperature (or other important climatic factors), and not only the mean values.
– It is not desirable to move material from a coastal climate to a continental one, or vice versa.

However, these recommendations are too general, and it is useful to combine information on ecological similarities among deployment and procurement zones, and from concrete transfer experiments. In general, there are site-site transfer rules (among planting sites and seed sources or stands), site-region transfer rules (among plantation sites and regions of provenance) or region-region rules (among deployment regions and regions of provenance). These types of rules are summarized in Table 5.

TABLE 5. Types of transfer rules of forest reproductive material.

Use of forest	Procurement of forest reproductive material	
Reproductive material	*Seed source, Stand or Tree*	*Region of provenance*
Plantation site	Site - Site	Site - Region
Deployment Region	-	Region – Region

Depending on the category of the reproductive material, different kinds of information can be used.

Source-identified and selected material. Usually there is ecological information at the region of provenance level, and some information on the performance of a limited number of provenances in a limited number of sites. We have to rely on the phenotypic characterization of the basic material and on the ecological similarities among planting sites and regions of provenance. The most common approach is to establish some relationship among regions of provenance and planting regions for the most important species (García del Barrio et al. 2001, CEMAGREF 2003). An example of these transfer rules is shown in Table 6. When information on the performance in provenance tests is available, it is possible to use this information to establish some predictive models (e.g., Hamann et al. 2000, Westfall 1992, Parker and Lesser 2004).

Qualified material. Usually there is information on the phenotypic characteristics of the individuals, the ecological characteristics of the sites where the materials were selected, and some information (derived from the empirical experience) on the performance of the materials. This material is not linked to a region of provenance, and we have to establish the similarity between the planting site and the site from where the material was selected.

Tested material. In this case, there is information on the performance of the material on different experimental sites, including the importance of the genotype-environment interaction. It is known the region of likely adaptation within the country in which the test was carried out and the characteristics which might limit its usefulness. These characteristics can help to determine the transfer guidelines of such material. This information is available for some of the species (e.g., Padró 1992, CEMAGREF 2003).

TABLE 6. Example of transfer rules for different species in Spain (Alía et al. 1999, García del Barrio et al. 2004). For each deployment zone, the codes of the most suitable regions of provenance and the level of recommendation, from low (light green) to very high (dark green), are included.

Deployment Zone	*Procurement zone (Region of Provenance)* *F. sylvatica*	*P. halepensis*	*P. nigra*	*P. pinaster*	*P. pinea*	*P. sylvestris*
1				1a		10
2	1			1b		10
3	2-4			1a		10
4	1-2-5		7			10
5	3-5		2-7	2-9		1-8-10
6	7-8			1a		2
7	5-6-8	4	3	3-9		2-3-4
8	9-11-12		2-3			3-4-5-6-7
9	10-12-13-14	3-4	1-2-3-4-5		7	3-4-7
10		1-2		6-9-C	6-7	
11		3-6	3-4-5	6-B	6-7	4-7-16
12		3-5-6-14				
13		5-9-14	7	10		14-15
14		4-6-9	5-7	9		4-8
15	16-17		3-10	9		8
16		9-14	5-7-10	8-9	1	8-10
17		9-14		2-8	1	
18				1a-4-6		10
19		14	8-9	6	2	10-11
20	18	14	8-9	6-7	2	9-10
21		5-7-9	7	11-12		12-14
22		7	7	12-13		12
…						

Conclusions

The genetic quality of the reproductive materials used in restoration programmes determines some of the future characteristics of the plantations, as variables related to adaptation, genetic diversity and growth have a strong genetic determinism.

The EU scheme on marketing of forest reproductive material is applied to 54 forest species, but the principles of such scheme are valid for other species relevant in restoration programmes. In those cases, it would be desirable to precisely define the type and characteristics of the basic material from which the reproductive material must be collected.

Therefore, when choosing reproductive material, we should take into consideration different factors as the characteristics of the basic material from which the material was collected, the category of the reproductive material, the region of provenance and origin of the material, and the information on the seed transfer rules available for the species or materials under consideration. Most of these factors are related to the future adaptability of the tree plantations, and many studies are now focused to provide information on such characteristics. The managers in charge of planning a restoration program must analyse the different materials available (species, regions, of provenance, type of basic material, category of the reproductive material) in order to select the most adapted to the planting conditions. It is necessary to take into consideration that this material can also affect the genetic diversity of the present and future forests.

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Assessing Morphological and Physiological Plant Quality for Mediterranean Woodland Restoration Projects

7

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Introduction

Planting seedlings grown in a nursery is often the main way for introducing or re-introducing plant species in woodland restoration projects. If species have been selected properly, the main factors affecting plantation success are environmental conditions, soil preparation and the quality of seedlings (South 2000).

Several reviews on plant quality have been published (see Ritchie 1984, Duryea 1985, Mattsson 1997, Wilson and Jacobs 2006), but most of the experience on this topic has been acquired from humid-temperate and boreal trees. Little information exists on plant quality of species from other biomes. Application of the experience gained in humid-temperate and boreal ecosystems to woodland species from other floras that have different phylogenetic background and environmental constraints than that of humid-temperate and boreal ecosystems must be done carefully.

During the last fifteen years there has been an increase in the study of plant quality, cultivation and plantation techniques of Mediterranean woody species. Our aim in this chapter is to review the procedures and importance for assessing forest plant quality, putting emphasis on recent experience gained with Mediterranean species and highlighting the differences observed *vis à vis* with humid-temperate and boreal forest species. As container stock is the predominant cultivation system of Mediterranean species in the European Union most of the information and examples given here will refer to this type of stock.

Concept of plant quality and importance of its assessment

Plant quality can be defined as the capacity of seedlings to survive and grow after transplanting in a specific environment (Ritchie 1984, Wilson and Jacobs 2005). The survival and growth capacity of a seedling depends on its carbon, water and nutrient economy, which are ultimately determined by the structure and physiological attributes of the plant (Burdett 1990). Plant functional attributes are genetically determined (see Chapter

6, this volume), but they are also plastic phenotypically and can vary depending on resource availability and the environment under which plants grow. The main aim of plant quality research is to determine which structural and physiological properties must seedlings attain for surviving and growing fast in a specific planting environment and how can these functional properties be achieved during nursery cultivation.

The functional attributes that promote seedling success in harsh sites are different than those in mesic sites (van den Driessche 1992). Thus, ideally, plant quality should be adjusted to the characteristics of planting site (Rose et al. 1990). Plant quality may vary over time. For instance, frost hardiness in most temperate species increases from the fall through the winter and then it decreases again through spring in response to changes in photoperiod and temperature (Grossnickle 1992). Thus, assessment of plant quality prior to planting must be done as closer as possible to planting date. Plant quality may also change with plant age. Nicolás Peragón (2004) observed that two-year old *Quercus faginea* seedlings had lower root growth capacity, survival and growth than one-year old plants.

The effect of planting poor quality stock can last for many years or it can be apparent only many years late after planting, as it occurs with root deformations that can reduce tree stability over the long-term (Lindström and Rune 1999).

Plant quality can be assessed by measuring several morphological and physiological attributes (material attributes), or by examining plant performance after subjecting them to specific environmental conditions (performance attributes). The final goal of testing plant quality prior to planting is to prevent plant lots with low survival and growth potential to be planted and to predict the *potential* out-planting performance of planted stock. Plant quality assessment cannot tell us the actual performances of seedlings because out-planting performance also depends on other factors some of which vary stochastically. Assessment of plant quality and use of high-quality plants are important because attainment of restoration and ecological objectives are secured or achieved faster and spread of diseases in natural populations is prevented. Similarly, the final economic cost of restoration projects may be increased if seedlings have to be replanted or if expensive post-planting cares, like irrigation, have to be used to warrant seedling establishment. Nurserymen should produce high-quality seedlings because it warrants consumer confidence and discards that planting failures are due to poor quality stock.

Assessment of plant physiological and performance attributes strongly increases the cost of a plant quality-testing program but it has saved a lot of money in some regions of North America. In British Columbia (Canada), plant quality assessment (physiological quality tests included) amounted 0.4% of the planting program cost (Dunsworth 1997). Assessment of physiological attributes in a plant quality-testing program is worthwhile when planting failures are frequent and planting and seedling cultivation costs are high. It might also be desirable when new species are used in reforestation programs whose biology, cultivation, and transplanting performance are largely unknown.

Material attributes. The morphological component of plant quality

The morphological quality of plants comprises a set of attributes that measure the structure, colour and appearance of the plant. Morphological attributes are the basis of the European Union legislation that regulates plant quality, although specific traits have only been defined for Mediterranean species (Directive 1999/105/CE). Morphological attributes can be either qualitative or quantitative, and most of them can be assessed with simple measurements.

Qualitative morphological attributes

In Table 1 are given the main qualitative morphological attributes used in plant quality assessment together with their rationale, and some cautions to be observed when these attributes are applied to Mediterranean species in a restoration context. The advantage of qualitative morphological attributes is that they are easy to assess. Their disadvantage is that they are to some extent subjective and have limited out-planting predictive capacity. Most qualitative morphological traits have been established from the cultivation and plantation experience accumulated in boreal and humid-temperate tree species. These species have been mainly planted for timber production so possession of specific traits important for timber quality, as single and straight stems, have considered desirable traits in seedlings. Application of certain classical qualitative attributes to shrubs and trees from other biomes or to stock used for restoration purposes might not be straightforward and some adjustment may be needed (Peñuelas and Ocaña 2000). For instance, multi-stem plants tend to be rejected because they produce low timber quality trees but in *Retama sphaerocarpa, a* Mediterranean leafless leguminous species, high-quality plants always produce multiple stems. Similarly, the presence of terminal buds is considered a sign of dormant and cold hardened state. However, in some species cold hardening is not associated with the presence of an apical bud (Colombo et al. 2001). Furthermore, many Mediterranean species have indeterminate growth and do not produce typical resting buds or if they form them they appear in saplings but not in seedlings.

Quantitative morphological attributes

Quantitative morphological attributes are measurements of the shoot and root size. Quantitative traits are used in both scientific studies on plant quality and in operational plant quality assessment. The root collar diameter (RCD), shoot height (from RCD to stem apex), shoot mass, and root mass are the most frequently measured attributes. The number of first order of laterals roots and root volume has also been used in bareroot stock quality assessment. From these measures several indexes have been developed. The most common are the shoot height/Root collar diameter ratio (i.e., shoot slenderness) and the shoot mass/root mass ratio (S/R) that is a proxy to the potential transpiration-water uptake balance of the plant (Ritchie 1984, Thompson 1985, Mexal and Landis 1990, Villar-Salvador 2003). Due to their measurement simplicity, shoot height and root collar diameter are the most

TABLE 1. Main qualitative morphological attributes used in plant quality assessment and their rationale. Some attributes are regulated by European legislation and the lots of plants with more than 5% of individuals with these traits must be rejected.

Attribute	Rationale	Comments on the application to Mediterranean species
Regulated by European Union legislation		
Injuries, except those caused during lifting or by pruning.	This attribute is especially important for bare-root stock. Injured plants have low vigour and poor establishment.	Injuries caused at lifting are rare in container plants but rough handling may damage plant during their transport. Pruning in Mediterranean container nurseries is not a common practice.
Lack of terminal buds	In many boreal and humid-temperate species, dormant and cold hardened seedlings form a terminal bud. Healthy apical buds produce well-developed shoots in spring.	Many Mediterranean species do not form a typical winter bud (a meristem protected by scales). This is the case of some junipers, *Arbutus unedo*, *Pistacia lentiscus*, *Viburnum tinus*, which have naked buds or meristems protected by leaves. Mediterranean pines do not form winter buds until they are saplings.
Multiple stems and main stem bifurcated	Multiple stems may indicate that two or more plants are growing in the same container cell or that the apical meristem bifurcated early, or resprouting has occurred. For timber production, it reduces growth and timber quality, and increases silviculture costs.	Many Mediterranean trees and shrubs have apical dominance as seedlings. However, some species, e.g. some evergreen oaks, can resprout if they are retained two or three years in the nursery. Whether this type of plant has lower out-planting performance is unknown. Some shrubs and many chamaephytes commonly produce multistem individuals.
Deformed root system	Spiralized roots or up growing roots in container plants may reduce new root egress and this may impair early establishment and future tree stability	
Wilted or chlorotic foliage or presence of rottenness or of any disease	Wilted or chlorotic foliage may indicate that the plant is diseased, has experienced severe drought or heat or has a deficient nutrient concentration. Planting infected plants can spread diseases or pests	
Unbalanced shoot root ratio (S/R)	Excessive S/R can cause water stress if new roots are not produced or soil is dry.	In dry areas low S/R is considered a desired trait. However, very low S/R may increase plant maintenance costs and therefore diminish growth
Other important qualitative traits not regulated by the European Union legislation		
Excessive stem curvature	Important for timber plantations because reduces the amount of profitable timber and its quality	Should be avoided if reduces growth and competitive capacity of seedlings
Actively growing shoots and unhardened plants	Plants with growing shoots are not hardened and consequently are less stress-resistant	Important for species planted in cold winter areas.
Lack of branches	The lack of lateral branches in some species may indicate high cultivation density or heavy shading during cultivation	Many species do not produce branches during the first year in the nursery
Poor developed plugs	For container stock poor developed plugs is a symptom of poor root development. These plants experience greater stress during their manipulation and plantation.	

commonly morphological quantitative attributes used for operational quality assessment. Many countries have regulated the shoot height and RCD standards for acceptable seedlings in many planted tree species. For instance, one-year old *Quercus ilex* (Holm oak) seedlings of acceptable quality in any of the Mediterranean countries of the European Union must have a shoot height ranging within 8 to 30 cm and a minimum RCD of 2mm.

Studies done with boreal and humid-temperate species show that plants size has a quite good predictive capacity of out-planting performance, which it frequently increases with shoot and root size, especially in mesic planting sites. Some experiments have shown that seedlings larger than conventional standards can be a promising alternative to herbicides because they compete better with weeds than the smaller conventional stock (Lamhamedi, *et al.* 1998). Relationships of survival and growth with S/R are less clear and often contradictory (Lopushinsky and Beebe 1976, Thompson 1985, Tuttle et al. 1988, Mexal and Landis 1990, van den Driessche 1992, Bayley and Kietzka 1997, South 2000, South et al. 2005). Root collar diameter and plant mass tend to predict better out-planting performance than shoot height and the relationship tends to be stronger with field growth than with survival. However, for plants of the same age there is a size and S/R top limit from which survival and growth plateau and decline (McDonald et al. 1984, Thompson 1985, South et al. 2005).

In Mediterranean environments, water stress is the main limiting factor for plant life and restoration success. This has conditioned Spanish foresters, which have traditionally preferred plants with small shoots and low S/R because they are considered to perform better in dry conditions than large plants and higher S/R (Royo et al. 1997) as they consume less water than plants with the opposite traits (Leiva and Fernández-Alés 1998). This sort of plant was produced with low amounts of fertilizer and frequently by restricting irrigation (Luis et al. 2004). Some evidences support that small plants with low S/R have greater survival than large plants with great S/R in Mediterranean dry areas. For instance, Trubat et al. (2003) observed that small seedlings had greater survival than large plants in *Pistacia lentiscus*. On the contrary, other authors have observed that large plants have higher survival and growth than small seedlings (Fig. 1) (Oliet et al. 1997, Luis et al. 2003, Puértolas et al.2003a, Villar-Salvador et al. 2004, Oliet et al. 2005, Tsakaldimi et al. 2005). A recent revision of 33 experiments on plant quality published by Spanish authors on Mediterranean species does not support the contention that the small seedlings with lower S/R have greater survival and growth than large seedlings and with greater S/R (Navarro et al. 2006). In most cases no relationship was observed between survival or growth and plant size or S/R. When survival was related to morphology, in most cases relationships with shoot size were positive while no trend could be concluded with S/R. Similarly, the results were independent of the type (woodland vs. abandoned cropland) and rainfall regime of the planting site. It was concluded that the present shoot size standards regulated by legislation in several Mediterranean woody trees should be higher (Table 2).

Material attributes. The physiological component of plant quality

Morphological attributes have limited predictive capacity of transplanting performance because they do not tell about the physiological status of the plant, which is also important for plant establishment. For instance, morphology will not indicate if non-structural carbohydrate reserves of a plant are low or its fine roots are damaged. Therefore, physiological quality attributes must be considered as a complement to morphological attributes rather than an alternative. Many physiological attributes have been utilised to assess plant quality (see Mattson 1997). We have focused on those most extensively studied and those that are most promising for plant quality assessment in Mediterranean species.

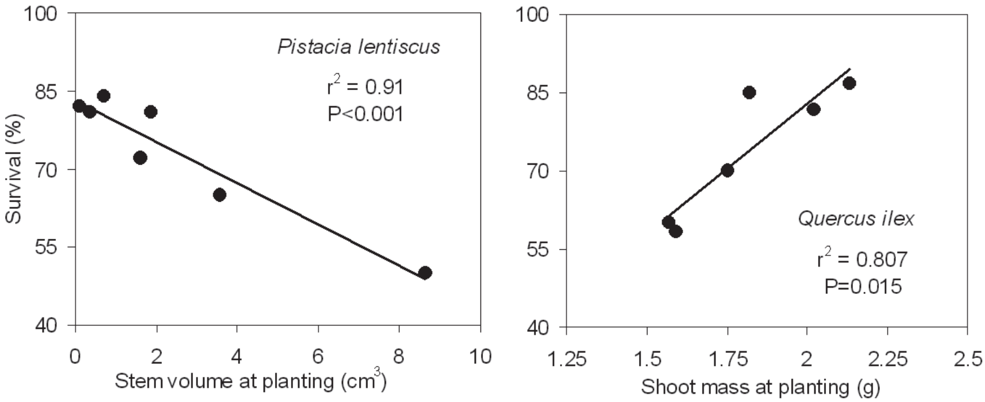

FIGURE 1. Relationship between survival and shoot size at time of planting in *Pistacia lentiscus* (left) and *Quercus ilex* (right). Figure of *P. lentiscus* was elaborated from data published in Trubat *et al.* (2003). Figure of *Q. ilex* is based on unpublished data of P. Villar-Salvador from an experiment with different fertilization and container treatments.

TABLE 2. Proposed standards of shoot height, diameter, slenderness and shoot /root ratios (S/R) for one-year old container seedlings in four common Mediterranean woody species. Values without brackets are proposed ranges and values in brackets are the standards recognized by European legislation. No legal standards exist for *Olea europaea*. (Data adopted from Navarro et al. 2006).

	Shoot height (cm)	Root collar diameter (mm)	Height / diameter ($cm\ mm^{-1}$)	S/R ($g\ g^{-1}$)
Pinus halepensis	15 - 30 *(10 - 25)*	3 - 4 *(>2)*	5-7	1.5 – 2.0
Pinus pinea	20-30 *(10 - 30)*	3.5 – 4.5 *(>3)*	5-7	2.0 – 2.5
Quercus ilex	20 - 30 *(8 - 30)*	4-5 *(>2)*	4-7	0.6 – 1
Olea europaea var. *sylvestris*	30 - 50	4-5	7-12	2-4

Mineral nutrient and non-structural carbohydrate concentration

Greater fertilization in the nursery enhances seedling growth and nutrient concentration, which frequently increases field performance (van den Driessche 1992, Villar-Salvador et al. 2004, Oliet et al. 2005). Nitrogen, phosphorus and potassium are the nutrients that most affect plant quality. Nitrogen is the most abundant macronutrient in the plant and it is strongly correlated to photosynthesis rate and growth. Plants remobilize N from old tissues to support new growth after transplanting. Therefore, plants with high N concentration compete better against weeds and have greater growth in oligotrophic soils (Salifu and Timmer 2003). Several studies have shown that post-planting survival and growth in Mediterranean species increases with tissue N concentration (Fig. 2a) (Oliet et al. 1997, Planelles 2004, Villar-Salvador et al. 2004). However, since morphology and tissue N are modified together by N fertilization in the nursery, it is difficult to disentangle the effect of plant size and N concentration on post-planting survival and growth. Plant N content (i.e., the product of dry weight nitrogen concentration by seedling dry weight) was better predictor of post-planting growth in *Pinus halepensis* than seedling dry weight alone (Fig. 2b and 2c). This illustrates that although seedling size determines post-planting growth, tissue N concentration also plays an important role.

Phosphorus forms part of ATP, certain enzymes, and membranes and is involved in the photosynthesis and respiration of the plant. Root growth is especially sensitive to P deficiencies. Effect of P deficiency on plant growth is less obvious than N deficiency. There are few studies linking tissue P to field performance but in semiarid leguminous species, transplanting survival increased with tissue P (Planelles 2004, Oliet et al. 2005).

Potassium is after N the most abundant nutrient in the plant. It regulates many metabolic functions like the osmotic adjustment, which has an important role in frost and water stress resistance. In spite of that, few studies have evidenced the effect of tissue K on transplanting performance (Christersson 1976).

Non-structural carbohydrates (TNC) comprise starch and a variety of soluble sugars, these latter having a prominent role in cold hardiness and drought tolerance of plants. TNC support respiration and growth especially when photosynthesis is low. Resprouting depends on TNC and in many deciduous species growth of new organs in spring relies on TNC (Dickson and Tomlinson 1996, McPherson and Williams 1998). TNC have proved to be important for transplanting performance in cold-stored seedlings when they deplete their TNC during storage and have low photosynthesis after transplanting (Puttonen 1986). We believe that poor transplanting performance in Mediterranean plantations due to low seedling TNC is probably not a common problem as seedlings rarely are cold-stored, container stock usually has lower transplanting shock than bareroot stock and most species maintain relatively high photosynthetic rates during winter (Villar-Salvador unpublished data).

Fine Root Electrolyte Leakage (REL)

A healthy and vigorous root system is essential for seedling establishment, especially in Mediterranean-climate regions, where seedlings need to extend rapidly their roots before the

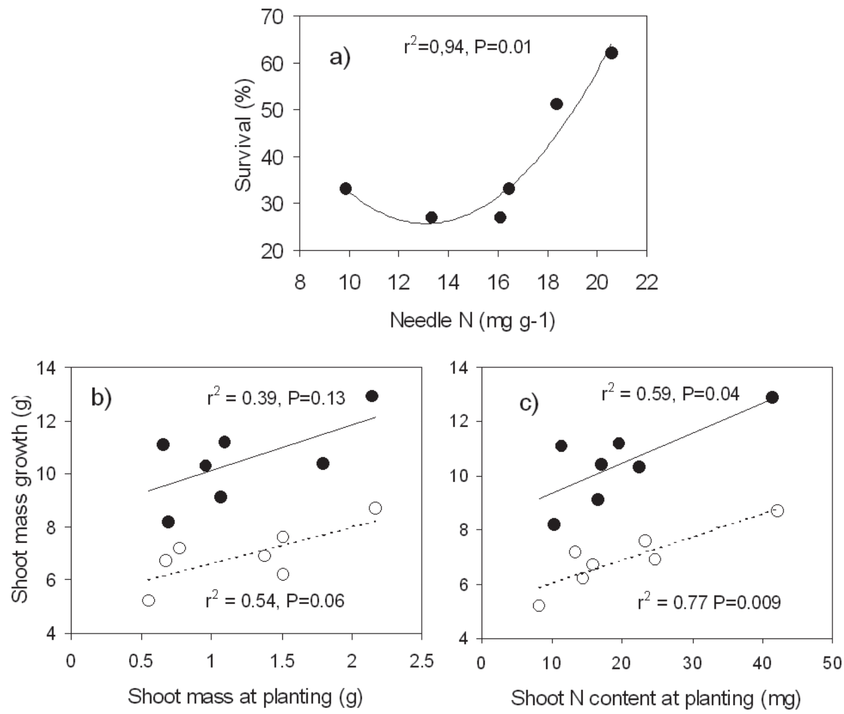

FIGURE 2. (a) Relationship between first year survival and foliar N concentration at planting in *Pinus halepensis* seedlings planted in a semiarid area in SE Spain (Modified from Oliet et al. 1997). Relationships between shoot mass growth two years after planting and shoot mass (b) and shoot nitrogen content (c) before planting in *Pinus halepensis* in two sites of contrasting stress conditions. Solid circles are results in an abandoned field with deep soils (mild stress site), and open circles are results in a slope with stony and shallow soils (high stress site) (Modified from Puértolas et al. 2003b).

onset of summer drought. McKay (1992) described a procedure to assess the vitality of fine roots based on the integrity of cellular membranes. Frost and desiccation can damage cell membrane causing the release ions outside the cell (electrolyte leakage). Electrolyte leakage usually is proportional to membrane damage and to stress intensity and it correlates well with seedling out-planting performance (McKay and White 1997). Electrolyte leakage can also be used to measure the plant's dormancy status. Root electrolyte leakage (REL) is used in plant quality control in the United Kingdom. Its methodology is simple and results can be obtained within 2 days.

Roots of containerised seedlings are sensitive to heavy frosts but damage is typically only detected after planting. In these cases REL may be a promising tool for early detection of frost-damaged plants in nurseries located in cold winter areas.

Performance attributes

Performance attributes are assessed by subjecting whole seedling to certain environmental conditions (optimal or not) and their growth, survival or any other physiological response is evaluated. The most frequently used performance tests are root growth capacity and frost hardiness (Dunsworth 1997).

Root Growth Potential Test (RGP)

It is the ability of a seedling to initiate and elongate new roots within a certain period of time (Ritchie 1985). A simple way to perform this test is to transplant seedlings to larger containers with peat, sand or perlite and placed in an optimum growing environment (wet and warm). This performance test has been used worldwide to assess plant quality due to its simplicity and because it measures the functional integrity and vigour of seedlings. Lots with damaged plants can be detected with this test (Fig. 3). RGP not only depends on the root physiological status, but also on the functional characteristics of the rest of the plant. For instance, RGP has been positively related with N concentration, seedling size, and frost resistance (Ritchie 1985, van den Driessche 1992, Pardos et al. 2003, Villar-Salvador et al. 2004).

From an operational point of view, RGP has two important disadvantages. On the one hand, RGP has important seasonal variations and it may vary depending on previous climatic conditions (Fernández and Pardos 1995). On the other hand, RPG test takes at least one week to be completed.

Although controversial, RGP is used to predict out-planting performance (see Simpson and Ritchie 1997). RGP tends to predict better absolute growth than survival (Fig. 3). Relationships between survival and RGP are frequently asymptotic, indicating that under a specific limit survival diminishes because seedlings are damaged or have low vigour. RGP predicts well the field survival and growth of seedlings in harsh sites where performances is tightly related to seedling vigour (Simpson and Ritchie 1997).

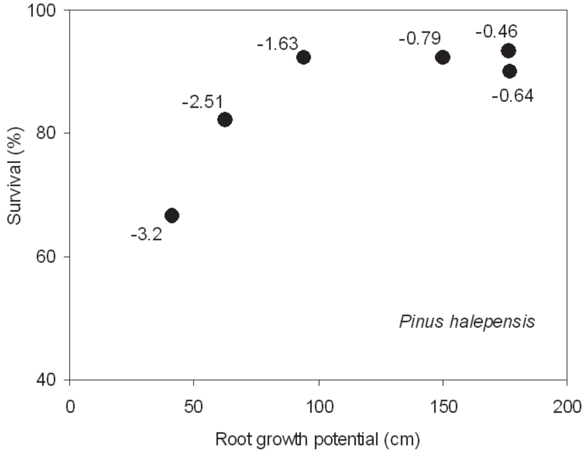

FIGURE 3. Relationship between out-planting survival and root growth potential in *Pinus halepensis*. Each point represents a treatment that experienced different pre-planting drought. For each point, predawn water potential (MPa) of each treatment before planting is represented. (Modified from Vallas-Cuesta et al. 1999).

RGP also predicts growth of vigorous seedlings in mild sites. In other cases, RGP may have limited predictive capacity. To increase the transplanting performance predictive capacity of RGP, some authors have proposed to carry out RGP tests under suboptimal conditions, similar to those that seedlings would encounter when planted (Folk and Grossnickle 1997).

Frost hardiness

Plants from temperate regions acclimate to frost during late summer and autumn. This process is called frost hardening or cold hardening and involves a number of biochemical and physiological changes, which allows plants to avoid freezing injury. Frost hardiness can be assessed subjecting whole seedlings or parts of them to artificial frost and evaluating frost damage. Temperature is reduced at a fixed speed to the target temperature. Cooling rate can determine the degree of injury (Sutinen et al. 2001), so it must be carefully controlled and the same rate used in the different tests. Frost damage can be evaluated at a unique freezing temperature or calculating the lethal temperature at which 50% of the seedlings are killed after subjecting different batches of seedlings to three or four decreasing freezing temperatures. Visual assessment and electrolyte leakage are the most common methods for evaluating frost damage. Visual assessment is simple and quantifies leaf and stem cambium necrosis or seedling mortality. However, it takes several weeks to be completed and requires a greenhouse or a growth chamber to maintain the seedlings under good environmental conditions after freezing. This limits their use for operational plant quality control, but it is a good option for research. The principles and procedures for electrolyte leakage determination are the same described for REL tests and its main advantage over visual damage assessment is its rapidity. However, before using electrolyte leakage in operational plant quality assessment it has to be calibrated for each species with visual frost damage determinations.

Frost hardening of forest species has been extensively studied in boreal and wet-temperate species. In cold climates, it is essential to have information about hardening and dehardening cycles of plants and how the nursery practices and environment influence this processes. In Mediterranean climates, where the main limitation for plant establishment is summer drought, researchers have focused more on plant water relations. However, cold is also important for plant life in Mediterranean climates, especially in inland and highland areas where frosts can last five or six months. There are few studies on frost hardiness of the Mediterranean forest species, yet the importance of low temperature on plant growth, survival, and drought hardiness has been demonstrated in *Q. ilex* and *P. halepensis* (Larcher and Mair 1969, Pardos et al. 2003, Mollá et al. 2006).

Factors that affect plant quality

Plant quality and out-planting performance depends on the environmental conditions and cultural practices in the nursery and how plant is handled after lifting and prior planting. In this section we briefly review these factors.

Fertilization

Fertilization in the nursery has a strong influence on plant morphology and physiology and its transplanting performance. Nitrogen is the most important nutrient and high N fertilization rates increase seedling growth, S/R, photosynthetic rate, N concentration and root growth capacity. Contrary to what many foresters and nurserymen assume, poor fertilization in Mediterranean species usually reduces their transplanting survival and growth (Oliet et al. 1997, Villar-Salvador 2004, Oliet et al. 2005). At present, we do not know yet which is the optimal tissue nutrient concentration for Mediterranean species. Good transplanting performance is obtained when N fertilization rate is greater than 70 mg $plant^{-1}$. However, plasticity of functional attributes in response to N fertilization is species-dependent, Mediterranean pines being more plastic than oaks (Oliet et al. 1997, Luis et al. 2003, Villar-Salvador et al. 2004).

The ideal amount of fertilizer to grow a seedling is that which maximises seedling nutrient loading and stress resistance without strong morphological imbalances. Excessive N fertilization may reduce frost and drought hardiness and transplanting performance (van den Driessche 1988, Colombo et al. 2001). In Mediterranean species, high N fertilization reduces frost hardiness in *Pinus halepensis* and *P. pinea*, but no effect was observed in *Juniperus thurifera* and *Quercus coccifera* (Puértolas et al. 2005, Villar-Salvador et al. 2005).

Some techniques have been developed to promote nutrient loading of cultivated seedlings without morphological imbalances. One of these techniques consists of maintaining or increasing fertilization after growth cessation in the autumn rather than restricting it, as it is commonly done. Nutrients provided during this period are not diluted with current growth and concentrate in the plant. In Mediterranean pines, late-season N fertilization increased field performance in *P. halepensis* (Puértolas et al. 2003a). However application of late-season fertilization in Mediterranean nurseries located in mild winter areas may be difficult as many species keep growing actively up to the end of the autumn. Exponential fertilization can be a more promising technique for nutrient loading. It is based on the “steady-state nutrition” concept (Ingestad and Lund 1986) and consists of the addition of high fertilizer inputs at exponential rather than constant rates, following seedling growth (Timmer and Aidelbaum 1996). There have been few experiments carried out on Mediterranean species (Carrasco et al. 2001) and much research on this subject is needed before it can be widely recommended.

Irrigation

Water is essential for many basic physiological processes of plants (Landis et al. 1989). Low watering reduces plant growth and nutrient uptake, and increases S/R. It can also cause upgrowing roots if the lower part of the plug remains dry, and salinization of the growing medium. On the contrary, water excess can induce loss of root vitality, proliferation of fungi diseases spread, impairment of root architecture and nutrient leakage. Watering regime must be adjusted through the cultivation and between species. Broadleaved species need special attention due to the high interception of water by leaves, which cause its irregular distribution among plants. In hot climates adequate irrigation is essential to prevent foliage overheating and consequently seedling damage.

Reduction of water supply during the late stages of nursery cultivation has been used to acclimate seedlings to water stress (drought hardening). Drought hardening increases the water stress resistance of plants but the type and magnitude of the response is species-dependent. In Mediterranean species, no clear trend of drought conditioning in the nursery on transplanting performance could be concluded (Vilagrosa et al. 2006).

Growing medium

The function of growing medium is to store water and nutrients that the plant can uptake and to anchor and maintain up straight the plant in the container. Therefore, growing medium has a profound influence on seedling morphology and physiology (Guehl et al. 1989). Peat is the most commonly used growing medium due to physical and chemical properties. Mixtures of this material with other organic and inorganic compounds (sand, perlite, vermiculite, pine bark, etc) are frequently used to improve its structural stability. One of the limitations of peat management is that it becomes hydrophobic if it dries excessively. This property has been considered to hinder seedling establishment in dry sites, although there are no studies that support this. Other growing medium alternatives like coconut fibre, ground pine cones and bark, or saw dust and wine distillery wastes have been tested (Landis et al. 1990). As peat has to be imported in most Mediterranean countries these alternative should be seriously considered, but studies about the effect of different growing media on Mediterranean species are almost inexistent (see Ruano et al. 2001).

Containers

Plant morphology depends on the dimensions of the container used and on cultivation density. Plant size tends to increase with container volume without great effects on S/R. High growing density increases plant height and S/R but reduces root collar diameter, plant mass and the number of lateral branches (Landis et al. 1990, Domínguez-Lerena et al. 2006). Tissue nutrient concentration increased with container volume in *P. pinea* and in general transplanting survival and growth in Mediterranean species tends to be higher when cultivated in large rather than in small volume containers (Domínguez-Lerena 1999, Domínguez-Lerena et al. 2006). Most Spanish nurseries grow Mediterranean species in containers with volumes larger than 200 ml. Stock cultivated in container volumes of 250-300 ml tend to have good transplanting results at an acceptable cultivation costs. Transplanting survival of species with strong taproots, such as oaks, increases when cultivated in deep containers than in shallow containers (Domínguez-Lerena 1999).

Root architecture before and after transplanting is also influenced by container structure. Containers may induce spiralized roots that may reduce out-planting performance and the future structural stability of trees due to poor lateral root egress (Lindström and Rune 1999). This can be prevented by cultivating seedlings in square shape containers or in containers with vertical internal ribs. Although they prevent root spiralling, vertical ribs cause a typical root conformation where lateral roots are forced to grow downward and are air-pruned at the drainage hole. As a consequence, the growth of new lateral roots after planting is concentrated in the lower part of the plug and the root system is far from natural, which may impair tree stability (Rune 2003). A solution is to prune lateral roots. This can be done by chemicals such as copper salts coated in the inside lateral walls of the container or by lateral air pruning, where lateral root tips contact air in vertical slits in the container wall. However this latter system is difficult to implement in nurseries located in hot places because desiccation of growing media is fast and plants need to be watered very frequently.

Seedling storage, rough handling, and nursery location

Seedling quality can be impaired during their storage prior to planting due to seedling desiccation, loss of sugar reserves, and mould development (McKay 1997). Desiccation is more likely to occur in bare-root stock than in containerised stock because the water stored in the plug buffers for desiccation. Therefore it is important to ensure the complete plug is well hydrated at lifting. As most Mediterranean species are cultivated in containers, plant desiccation is not a frequent problem. However, desiccation can occur if plants are stored for prolonged periods at the planting site. *Pinus halepensis* seedlings reduced their transplanting performance when plants desiccated to predawn water potential < -2MPa (Vallas-Cuesta 1999). Prolonged storage of plants in darkness, even at low temperature, can reduce carbohydrate reserves and this can impair transplanting performance (Puttonen 1986). Plant storage in darkness and at low temperature is not a frequent practice in Mediterranean countries because winter is not as cold as at higher latitudes, planting is concentrated in the fall and winter, and it increases cultivation costs.

Plants can be damaged due to rough handling during lifting, transport and planting. Particular care must be taken when plants are loaded and unloaded from vehicles and distributed in the field (McKay 1997).

Finally, nursery location can affect plant quality due to differences in winter temperature among nurseries. Seedlings that are cultivated in nurseries located at higher altitude and latitude are more cold-resistant and harden earlier than seedlings cultivated in nurseries placed at coastal or low latitude sites (Mollá et al. 2006). This is important if stock has to be planted in cold sites.

Concluding remarks

Utilisation of high-quality seedlings is important for revegetation success. Cultivation conditions in the nursery and plant care before planting determine the quality of planted seedlings. Plant quality can be assessed by simple morphological attributes that measure the structure, colour and appearance of the plant. However, morphological attributes have limited capacity to discriminate poor quality plants, being recommendable to complement them by assessing physiological attributes relevant for plant establishment and/or assessing the performance of seedlings to specific environmental conditions.

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

8

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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 (μmol 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 transpiration at high water availability levels (Valladares and Pugnaire 1999, Ramírez et al. 2006, 2008). 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 2008). 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). However, the role of these roots in capturing soil resources is still under discussion (Puigdefábregas and Sánchez 1996, Ramirez et al. 2008).

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, Bautista et al. 2007), 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 et al. 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.

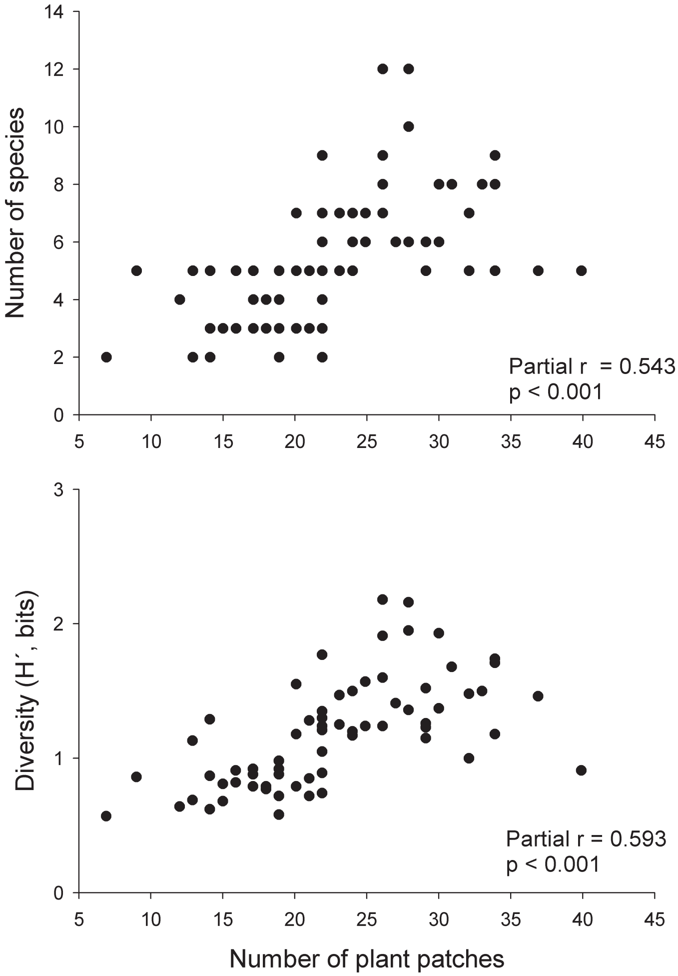

FIGURE 1. Relationships between the number of discrete plant patches and the richness and diversity of perennial vascular plants in semi-arid steppes of SE Spain. Results of partial correlation analyses, where the effect of plant cover is controlled, are shown in the lower right margin of each graph. 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 et al. 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.

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

They 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).

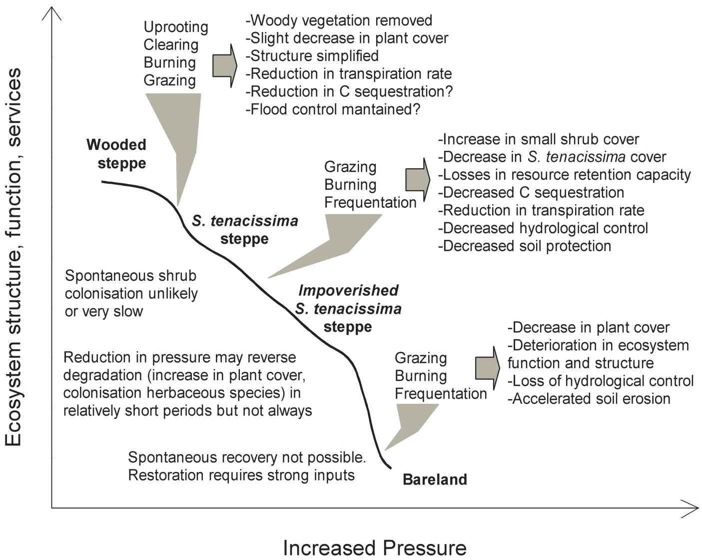

FIGURE 2. A schematic view of *S. tenacissima* degradation and aggradation dynamics in response to external anthropogenic pressures. Direction and intensity of changes in ecosystem structure, function and services will be highly dependent on climate, topography, lithology, and on the variable measured (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 20^{th} century. Restoration of semi-arid steppes in SE Spain, historically focused on hydrological control (e.g., Sierra Espuña, Murcia 1886; see Chapter 4, 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 20^{th} 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 re-distibution), 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 mycrorrhizae 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 interannual 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 climatic 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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Runoff and Erosion from Wildfires and Roads: Effects and Mitigation

9

Lee H. MacDonald and Isaac J. Larsen

Introduction

Forest disturbance can lead to land degradation, particularly in drier areas that are more sensitive to desertification. Both unpaved forest roads and high-severity forest fires can increase runoff and erosion rates by one or more orders of magnitude relative to undisturbed forested areas, and these can have long-term adverse effects on site productivity, water supplies, and other downstream resources. Forest managers commonly apply emergency rehabilitation treatments after wildfires to reduce runoff and erosion, but there are relatively few data rigorously testing the effectiveness of such treatments. Even fewer studies have compared long-term erosion and sediment delivery rates from roads and wildfires, yet such information is urgently needed to guide forest management.

Undisturbed forests typically have high infiltration rates (>50 mm h^{-1}) and very little bare soil (Robichaud 2000, Martin and Moody 2001, Libohova 2004). The high infiltration rates mean that nearly all of the precipitation and snowmelt infiltrates into the soil. Hence water flows to the drainage network primarily by subsurface pathways, resulting in low peak flows (Hewlett 1982, MacDonald and Stednick 2003), very low surface erosion rates and sediment yields (typically 0.005-0.5 Mg ha^{-1} yr^{-1}) (Patric et al. 1984, Shakesby and Doerr 2006), and runoff that is very high in quality and useful for municipal water supplies (Dissmeyer 2000).

Disturbances such as roads hinder infiltration and can serve as pathways for delivering water and sediment to streams, lakes, and wetlands (Trombulak and Frissell 2000). The low infiltration rates on unpaved road surfaces cause the dominant runoff process to shift from subsurface stormflow to infiltration-excess or Horton overland flow (HOF) (Robichaud et al. 2008). The low infiltration rates and high overland flow velocities greatly increase the size of peak flows and surface erosion rates (Dunne and Leopold 1978). Furthermore, unless a road is outsloped, the runoff and sediment from unpaved road segments often is concentrated in rills or ditches and directly routed to the stream channel network (Robichaud et al. 2008). In forested areas the human-induced increases in sediment loads are typically the pollutant of greatest concern (MacDonald 2000).

Wildfires are the other disturbance in forested environments that can greatly increase runoff and erosion rates. In many areas the risk of wildfires has increased as a result of human-induced changes in vegetation density, vegetation type, and the number of ignitions. Recent studies show that climate change also is increasing the risk of wildfires (Ryan 1991, Mouillot et al. 2002, Westerling et al. 2006). High-severity fires are of particular concern because they completely consume the surface organic layer (Neary et al. 2005a) and can induce a water repellent layer at or near the soil surface (DeBano 2000). Raindrop impact on the exposed mineral soil can detach soil particles and induce soil sealing, which reduces the infiltration rate. The resultant surface runoff greatly increases erosion rates by sheetwash, rill, and channel erosion (Shakesby and Doerr 2006). The change from subsurface to surface runoff and the loss of surface roughness greatly increases runoff velocities, and this further increases the size of peak flows and surface erosion rates. The risk of high runoff and erosion rates is substantially lower in areas burned at low or moderate severity because the fire does not consume all of the surface organic matter (Ice et al. 2004, DeBano et al. 2005).

Post-fire rehabilitation treatments –such as seeding and mulching– are commonly applied to severely-burned areas to reduce post-fire runoff and erosion. These treatments can be very costly, especially for large wildfires. For example, U.S. $72 million was spent on post-fire rehabilitation treatments after the 2000 Cerro Grande fire in New Mexico, and $25 million was spent after the 2002 Hayman fire in Colorado (Morton et al. 2003, Robichaud et al. 2003). The problem is that there are few data on the effectiveness of these treatments in reducing post-fire runoff and erosion (Robichaud et al. 2000, GAO 2003).

For the last six years we have been intensively studying how unpaved roads, wildfires, and post-fire rehabilitation treatments affect runoff and erosion rates in the Colorado Front Range, and the delivery of this sediment into and through the stream network. Much of this concerns stems from the fact that the South Platte River watershed provides 70% of the water for approximately two million people living in and around Denver, and both the quantity and the quality of this water is highly dependent on forest conditions and forest management activities. The specific objectives of this chapter are to: 1) summarize the effects of roads and fires on runoff and erosion in forested areas; 2) present our methods for measuring runoff and erosion so that they can be applied elsewhere; 3) review and explain the effectiveness of different post-fire rehabilitation treatments; and 4) compare the long-term erosion rates from unpaved forest roads and wildfires. By combining our detailed, process-based understanding with results from other areas, the information being presented is more broadly applicable. Both the methods and the results provide useful insights and guidance to other researchers as well as land managers.

Background

Effects of roads on runoff and erosion

In the absence of burning, unpaved roads are the dominant sediment source in forested areas (Megahan and King 2004). Infiltration rates for compacted road surfaces are typically 0.1 to

5 mm h^{-1}, and these low rates mean that rainstorms and snowmelt can generate overland flow on the road surface (Robichaud et al. 2008). Roads that are cut into the sideslopes can intercept the downslope subsurface water flow, and the conversion of subsurface to surface flow further increases the amount of road runoff and the size of peak flows (e.g., Wigmosta and Perkins 2001, Wemple and Jones 2003). The lack of surface cover exposes the road surface to rainsplash erosion, and the high runoff rates subject the road surface to sheetwash and rill erosion. Road grading and vehicular traffic generally increase road erosion rates, as these increase the supply of easily erodible sediment (Reid and Dunne 1984, Luce and Black 2001, Ramos-Scharrón and MacDonald 2005).

The runoff and erosion from unpaved roads may have little effect if these materials are discharged in a diffuse manner onto undisturbed hillslopes where infiltration rates are high and the sediment is deposited or captured by litter, downed logs, and vegetation. On the other hand, road segments that cross perennial or ephemeral streams can deliver water and sediment directly to the stream. The amount of runoff and sediment that is delivered to streams from these other road segments depends on the distance between the road and the stream, the hillslope gradient, the infiltration rate and surface roughness in the area between the road and the stream, the amount of runoff, and whether the road design disperses or concentrates road surface runoff (e.g., Megahan and Ketcheson 1996, Croke and Mockler 2001). A compilation of studies shows that the proportion of roads that are connected to the stream network is a linear function of the mean annual precipitation (Fig. 1). In the absence of local data, the relationship shown in Figure 1 can be used to estimate the proportion of unpaved roads that are likely to be delivering runoff and sediment to the stream channel network.

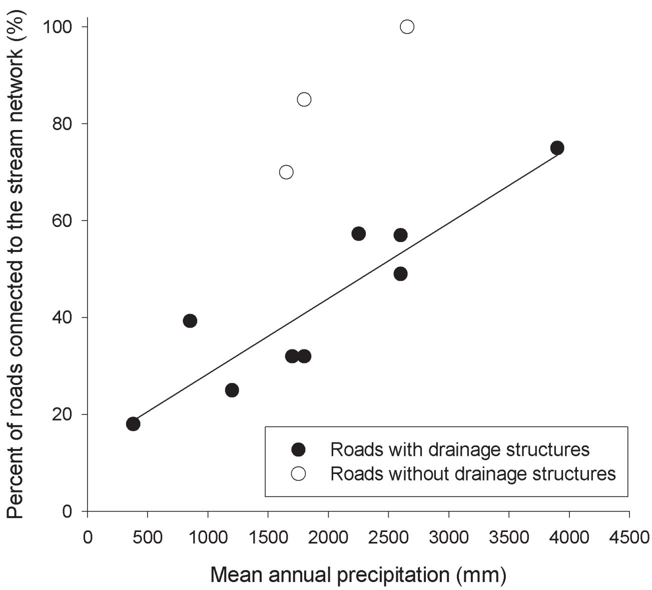

FIGURE 1. Percent of roads connected to the stream network versus mean annual precipitation for roads with and without engineered drainage structures. Regression line is for roads with engineered drainage structures (from Coe 2006).

Effects of forest fires on soils, runoff, and erosion

High-severity wildfires consume all of the surface organic matter and expose the underlying mineral soil (Neary et al. 2005a). In most coniferous forests and other vegetation types such as matorral, fynbos, and chaparral, the burning litter vaporizes water repellent compounds that are forced downwards by the heat of the fire. These compounds condense on the underlying, cooler soil particles, and they can induce a water repellent layer at or beneath the soil surface (Letey 2001). The depth of this water repellent layer increases with increased soil heating, and coarse-textured soils are more susceptible to the formation of a water-repellent layer than fine-textured soils because of their lower surface area (Huffman et al. 2001, DeBano et al. 2005). The water repellent layer is of concern because it can severely reduce infiltration rates and induce overland flow (Letey 2001, Benavides-Solorio and MacDonald 2001, 2002).

In moderate and high severity fires the loss of the protective litter layer exposes the mineral soil to rainsplash erosion. High severity fires also may burn the organic matter in the uppermost layer of the mineral soil, and the resulting loss of soil aggregates can greatly increase the soil erodibility (DeBano et al. 2005). The soil particles may clog the surface pores and induce surface sealing, which will further decrease infiltration rates (Neary et al.

FIGURE 2. View of a convergent hillslope in July 2002, just a few weeks after the 2002 Hayman wildfire. The metal rebars in the middle of the picture are the remants of a sediment fence that was installed before the wildfire. Prior to burning there was not a defined channel, and the first storms after the fire incised a channel that extends to within a few meters of the ridgetop.

1999). The loss of surface roughness by burning increases the velocity of surface runoff, and the combination of reduced infiltration and high overland flow velocities can increase the size of peak flows by one or two orders of magnitude (i.e., 10-100 times) (Scott 1993, Moody and Martin 2001a, Neary et al. 2005b).

In low severity fires not all of the surface organic material is burned. Because the soils do not become water repellent and the mineral soil is not directly exposed to rainsplash or soil sealing, low severity fires typically have little or no effect on infiltration and surface erosion rates (Robichaud 2000, Benavides-Solorio and MacDonald 2005).

The increase in erosion rates after high severity fires can be even greater than the increase in the size of peak flows because of the loss of soil aggregates and the exposure of the soil to rainsplash, sheetwash, and rill erosion (Neary et al. 1999, Moody et al. 2005). The lack of surface roughness results in high overland flow velocities, and this further increases the detachment and transport of soil particles. Rills and gullies readily form where the surface runoff becomes concentrated by topography, rocks, or logs. Rill and gully erosion (Fig. 2) can account for about 80% of the sediment generated from high-severity wildfires (Moody and Martin 2001a, Pietraszek 2006).

The net effect is that high-severity fires can increase sediment yields by two or more orders of magnitude (Robichaud et al. 2000, DeBano et al. 2005, Shakesby and Doerr 2006). The delivery of this sediment to downstream areas leads to channel aggradation and adverse effects on aquatic habitat and reservoir storage (Moody and Martin 2001a, Rinne and Jacoby 2005). Water quality is severely degraded by the high concentrations of ash and fine sediment, and fires also can result in high concentrations of nutrients and heavy metals (Neary et al. 2005c).

Over time the fire-induced soil water repellency breaks down and plant regrowth provides a protective cover of vegetation and litter (e.g., Robichaud and Brown 1999, MacDonald and Huffman 2004; Benavides-Solorio and MacDonald 2005). Runoff and erosion rates usually return to background levels after several years, but post-fire recovery can occur within three months or require up to 14 years (Shakesby and Doerr 2006). Recovery is more rapid as fire severity decreases (Pietraszek 2006).

Post-fire rehabilitation treatments

The adverse effects of high-severity fires on runoff and erosion rates often compel land managers to apply emergency rehabilitation treatments. These emergency rehabilitation treatments are designed to either increase revegetation rates and surface cover (e.g., seeding, mulching), or provide physical barriers for trapping runoff and sediment at the hillslope or watershed scale (e.g., contour log erosion barriers, check dams).

The most common post-fire rehabilitation treatments are grass seeding, mulching, and the placement of contour-felled logs (Robichaud et al. 2000, Raftoyannis and Spanos 2005). Grass seeding has been the most widely used technique because it is relatively inexpensive and can be rapidly applied over large areas by aircraft. Mulch immediately increases the amount of surface cover, but it is more difficult and costly to apply. The application of straw mulch also raises concerns about the possible introduction of weeds or other non-native species (Kruse et al. 2004, Keeley et al. 2006).

Contour-felled logs, or contour log erosion barriers, are burned trees that are cut down, de-limbed, and staked parallel to the contour on burned hillslopes. They are designed to trap the runoff and sediment coming from upslope areas. To be effective, a small trench needs to be dug upslope of the log and the excavated material has to be packed underneath the log to prevent underflow. The trench may temporarily enhance infiltration by cutting through the water repellent layer, and the trench also can slightly increase the water storage capacity on the hillslope (Wagenbrenner et al. 2006). Straw wattles and straw bales also are used to trap runoff and sediment from burned hillslopes (Robichaud 2005).

Monitoring methods

Monitoring the effects of fires and roads on soils, runoff, and erosion can be done at different spatial scales for different purposes. At the point or very small plot scale infiltration rates can be measured by minidisk (Lewis et al. 2006) or ring infiltrometers (Martin and Moody 2001a), but it is difficult to extrapolate these small-scale data to hillslopes or small catchments.

Soil water repellency can only be measured at the point scale, and this is most commonly done by measuring the Water Drop Penetration Time (WDPT). In this test one or more drops of water are placed on the soil surface and the time required for the water to penetrate the soil is recorded (Letey 1969). An alternative method is the Critical Surface Tension (CST), and this uses varying concentrations of ethanol in water. Higher ethanol concentrations lower the surface tension of water, and the CST is the surface tension of the drops that infiltrate the soil within 5 seconds (Watson and Letey 1970). Longer WDPT penetration times and lower CST values denote stronger water repellency. Though WDPT is more widely used than the CST, the CST procedure is faster, has less spatial variability, and has shown better correlations with predictive variables (Scott 2000, Huffman et al. 2001).

Changes in soil structure, cohesion, and erodibility can be assessed by measuring aggregate stability and critical shear stress (e.g., Badìa and Martì 2003, Mataix-Solera and Doerr 2004, Moody et al. 2005). The infiltration rates and soil conditions on unpaved roads can be readily compared to values from adjacent undisturbed sites, and this allows one to estimate the local effects of unpaved roads. Pre-burn data are almost never available for wildfires, and in larger fires there may be no immediately adjacent unburned sites to serve as reference conditions. These limitations make it more difficult to rigorously evaluate the effects of burning on soil properties as compared to the effects of unpaved roads.

Runoff and sediment yields can be measured at the plot scale ($\leq$~300 m^2) by capturing the overland flow produced by natural storms in containers. Rainfall simulations provide a more controlled means for assessing the effect of site characteristics and rainfall rates on runoff and erosion (e.g., Benavides-Solorio and MacDonald 2001 and 2002, Cerdà and Doerr 2005). Practical considerations usually limit rainfall simulations to plots that are 1 m^2 or smaller, although some studies have used plots of 10-300 m^2 (e.g., Wilson 1999, Johansen et al. 2001, Rulli et al. 2006).

At the hillslope and road segment scale, sediment production rates can be readily measured with sediment fences (Fig. 3). These are inexpensive and relatively simple to construct (Robichaud and Brown 2002; http://www.fs.fed.us/institute/middle_east/platte_pics/silt_fence.htm). Sediment fences need to be regularly checked and manually emptied in order to obtain valid data. Runoff can be measured at the hillslope scale by installing small flumes or weirs with water-level recorders, but these are much more costly than sediment fences.

Runoff and sediment yields are much more difficult and costly to measure at the watershed scale than at the plot or hillslope scale (Shakesby and Doerr 2006). Runoff can be most accurately measured by installing a flume or weir. The use of a standard design, such as a 90° V-notch weir or a Parshall flume, is advantageous because of the known relationships between water height and discharge. Measuring discharge in natural channels is more difficult because one must make the necessary field measurements to

FIGURE 3. A pair of sediment fences used for measuring hillslope-scale sediment yields after a wildfire.

establish the relationship between water level and streamflow, and these are less accurate and difficult to obtain at high flows (e.g., Kunze and Stednick 2006). Sediment yields can be measured at the watershed scale by constructing sediment rating curves from simultaneous measurements of streamflow and suspended sediment and/or bedload, or by trapping the eroded sediment behind debris dams (e.g., Rice et al. 1965, Moody and Martin 2001a). Measuring runoff after high-severity fires is extremely difficult because the high sediment yields tend to clog up flumes, fill the ponded area behind weirs, and alter the stage-discharge relationship by altering the channel cross-section through aggradation and/or incision. It also is much more difficult to replicate or compare sites at the watershed scale.

In summary, small-scale measurements are cheaper, more easily replicated, and can be used to isolate the effects of specific site conditions. Larger-scale measurements integrate much of the smaller-scale spatial variability and are closer to the scale of interest to land managers. The disadvantages of larger-scale measurements include their higher cost, the difficulty of replication, the difficulty of characterizing larger and more diverse areas, and the associated difficulty of making process-based interpretations of larger-scale data.

New insights from the Colorado Front Range

Road erosion

In the Colorado Front Range we have been measuring road erosion rates and assessing the connectivity between roads and streams since summer 2001. The most complete erosion

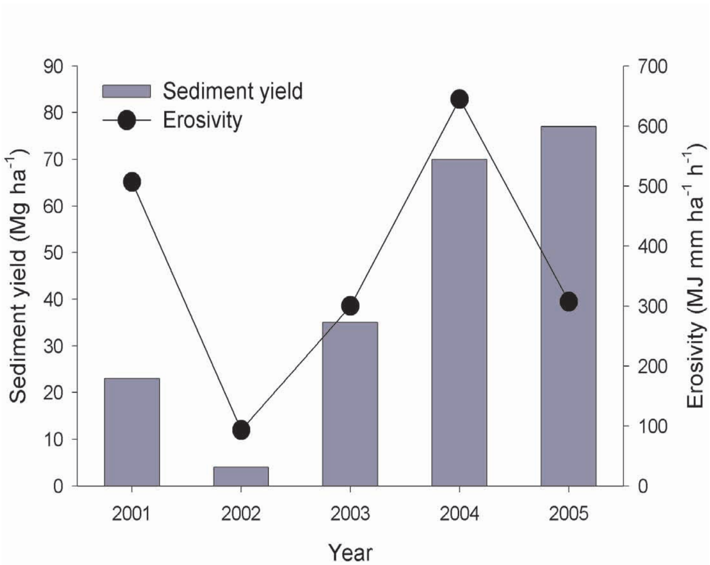

FIGURE 4. Mean annual sediment production and rainfall erosivity from 2001 to 2005 for eleven road segments along the Spring Creek road in the Upper South Platte River watershed in Colorado.

data are for five years from 11 unpaved road segments along the Spring Creek road in the Pike National Forest approximately 65 km southwest of Denver. From 2002 to 2006 the mean annual sediment production rate was 42 Mg per hectare of road surface. The importance of longer-term measurements is shown by the 10-fold variation in annual sediment production (Fig. 4). The high interannual variability is attributed primarily to the differences in rainfall erosivity, although the higher sediment yields in 2005 also may be due to an increase in traffic as a result of forest thinning operations.

Since unpaved roads occupy about 0.003% of the South Platte watershed, unpaved roads produce about 0.13 Mg ha^{-1} of sediment per year. Detailed surveys of 13.5 km of unpaved roads indicate that about 2.4 km or 18% of the roads drain directly to perennial or ephemeral streams via stream crossings, rills, or sediment plumes (Libohova 2004). This value is consistent with the relationship shown in Figure 1.

Surface cover, soil water repellency, runoff, and sediment yields for undisturbed vs. severely-burned hillslopes

Undisturbed ponderosa pine forests in Colorado typically have at least 85% surface cover and infiltration rates in excess of 100 mm h^{-1} (Martin and Moody 2001, Libohova 2004). These characteristics mean that overland flow is rare and surface erosion rates are very low (Morris and Moses 1987, Libohova 2004). We have collected over 100 hillslope-years of data from 34 undisturbed sites, and only one site with an unusually low amount of surface cover (<55%) has generated measurable amounts of sediment. No sediment was generated from any of the other sites, even though some sites with slopes of up to 55% have been subjected to rainfall intensities of more than 60 mm hr^{-1}. Similarly, no sediment has been produced from any of the hillslopes where over half of the trees were mechanically chipped to reduce wildfire risk (Libohova 2004).

As in many coniferous forests, the soils in the undisturbed ponderosa pine forests are water repellent at depths of 0-3 cm. Below 6 cm the soils in our study areas exhibit little soil water repellency (Libohova 2004). Burning at high and moderate severity strengthens the soil water repellency at 0 and 3 cm, and induces moderate to strong soil water repellency at a depth of 6 cm (Huffman et al. 2001). Multivariate analyses show that soil water repellency strengthens with increasing burn severity and sand content, and decreases with increasing soil moisture content (Huffman et al. 2001). In general, however, soil water repellency is highly variable in time and space (Doerr et al. 2008), and we found that the three predictive variables of burn severity, sand content, and soil moisture could only explain 30-41% of the variability in soil water repellency measured on two wild and three prescribed fires (Huffman et al. 2001).

Measurements over time indicate that the fire-enhanced soil water repellency in the Colorado Front Range is relatively short-lived. In the case of the Bobcat fire there was a significant decline in soil water repellency within three months, and the effect of burning on soil water repellency was not statistically detectable within 12 months (MacDonald and

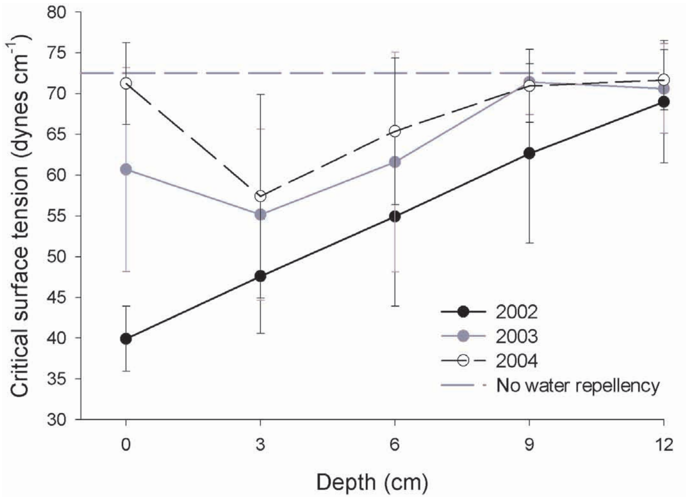

FIGURE 5. Mean soil water repellency over time at the Hayman fire using the CST procedure. Higher values indicate weaker soil water repellency, and the bars indicate one standard deviation.

Huffman 2004). Similarly, the soil water repellency was strongest at the soil surface and decreased with depth after the 2002 Hayman wildfire, but by the second year after burning this water repellency was not statistically significant compared to unburned sites (Fig. 5) (MacDonald et al. 2005). The greater persistence of soil water repellency at a depth of 3 cm relative to the soil surface may be due to the preferential erosion of water repellent particles and the faster chemical and physical breakup of the water repellent layer at the soil surface by solar radiation, biological activity, and freeze-thaw processes. Most other studies also have shown a relatively rapid decay of fire-induced soil water repellency (e.g., Hubbert et al. 2006; Doerr et al. 2008).

As soils wet up they no longer are water repellent (Leighton-Boyce et al. 2003, Hubbert and Oriol 2005). The soil moisture threshold for the shift from water repellent to hydrophilic appears to increase with increasing burn severity (MacDonald and Huffman 2004). For unburned sites adjacent to the Bobcat fire in Colorado there was no evidence of soil water repellency once the soil moisture content exceeded 10%. For burned sites the soil moisture threshold was 13% for sites burned at low severity, while sites burned at high severity could still be water repellent when the soil water content was 26% (MacDonald and Huffman 2004). In a California chaparral watershed the proportion of the surface with high or moderate water-repellency dropped from 49% to 4% when the soil moisture content reached 12% (Hubbert and Oriol 2005). These results and other studies indicate that post-fire soil water repellency is unlikely to increase runoff rates once the soils have wetted up, but soil water repellency can be re-established once the soils dry out (Leighton-Boyce et al. 2003).

Measurements at the small catchment scale in Colorado indicate that overland flow is initiated from severely burned areas when the maximum 30-minute rainfall intensity (I_{30}) exceeds about 7-10 mm h^{-1} (Moody and Martin 2001b, Kunze and Stednick 2006). Peak flows increase exponentially as I_{30} exceeds 10 mm h^{-1} (Moody and Martin 2001b), and the maximum peak flows of 4 to 24 m^3 s^{-1} km^{-2} from the Front Range of Colorado are comparable to the range of values (3.2-50 m^3 s^{-1} km^{-2}) measured from severely-burned areas in the western U.S. (Moody and Martin 2001a, b, Kunze and Stednick 2006). In the ponderosa pine zone in Colorado the post-fire increases in the size of peak flows and surface erosion rates persist for 2-5 years after a high-severity wildfire (Moody and Martin 2001a, Benavides-Solorio and MacDonald 2005, Kunze and Stednick 2006). Since the decrease in post-fire soil water repellency is much more rapid than the decrease in post-fire runoff and erosion rates, there must be some other process, such as soil sealing, that is contributing to the observed, longer-term increases in post-fire runoff and erosion.

Effects of fires on hillslope-scale sediment yields

Hillslope-scale sediment yield data have been analyzed from six Colorado fires (Benavides-Solorio and MacDonald 2005). Over 90% of the sediment was generated by high intensity summer thunderstorms. Very little sediment was generated by snowmelt because the soils were not repellent due to the wet conditions and snowmelt rates are much lower than the rainfall intensities for the larger summer thunderstorms.

The range of sediment production rates after fires as measured by sediment traps is from 0 to 70 Mg ha^{-1} yr^{-1}. The mean annual sediment production for high severity sites in the Bobcat fire was 8.7 Mg ha^{-1} for the first two years after burning, while the mean value for sites burned at moderate and low severity was less than 0.3 Mg ha^{-1} yr^{-1} (Fig. 6) (Benavides-Solorio and MacDonald 2005). The high severity sites in prescribed fires produced only about 10% as much sediment as the high severity sites in the Bobcat wildfire (Fig. 6), and this is attributed to the more patchy nature of the prescribed fires and greater surface cover in the prescribed fires due to litterfall and more rapid vegetative regrowth (Benavides-Solorio and MacDonald 2005).

Multivariate analyses showed that the amount of bare soil explained nearly two-thirds of the variability in annual sediment yields from the hillslope-scale plots in the Bobcat fire (Fig. 7). The lower sediment production rates in 2000, which was the year of burning, are due to the lack of large storm events. In summer 2001 there were more large storm events, and annual sediment yields were consistently high when there was more than about 35% bare soil (i.e., less than 65% surface cover). The same general trends were shown for a much larger data set by Pietraszek (2006), and studies in other areas also have documented the importance of surface cover in reducing runoff and erosion from forests and shrublands (e.g., Lowdermilk 1930, Brock and DeBano 1982, Robichaud and Brown 1999). The implication is that the progressive decline in post-fire sediment yields over time largely depends on the regeneration of surface cover.

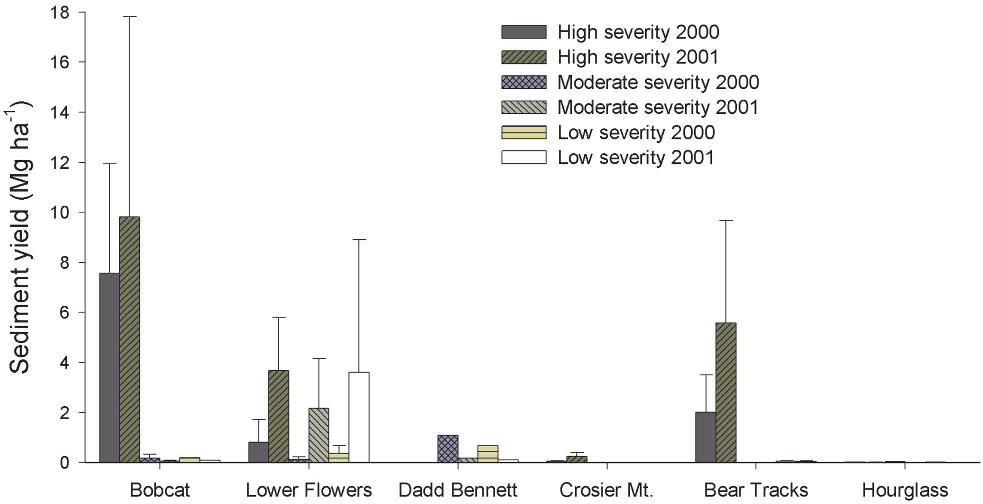

FIGURE 6. Sediment yields by burn severity for six Colorado fires for June-October 2000 and June-October 2001. Bars indicate one standard deviation. Not all severity classes were present in each fire.

After the amount of surface cover, the most important factors for predicting post-fire sediment yields in the Colorado Front Range are rainfall erosivity, soil texture, and fire severity (Pietraszek 2006). Rainfall erosivity is the most important of these additional factors, and its influence is greatest in recently-burned areas with little surface cover. Coarser soils tended to have lower sediment yields, and this can be attributed to the greater difficulty in detaching and transporting larger particles. Fire severity is a significant variable primarily because the amount of surface cover decreases with increasing severity. A multivariate model using percent bare soil, rainfall erosivity, soil texture, and fire severity explained 77% of the variability in post-fire sediment yields in the Colorado Front Range (Benavides-Solorio and MacDonald 2005).

The 2002 Hayman wildfire provided a unique opportunity to evaluate the effects of high-severity wildfires because it burned 20 study sites that had been established in the previous summer to evaluate the effects of a proposed forest thinning project. Prior to burning the mean amount of surface cover on each of these convergent hillslopes was about 85%, there were no channels or visual evidence of overland flow, and there were no measurable amounts of sediment in any of the sediment fences. After burning the mean amount of surface cover dropped to less than 5%, and the first rainstorm of only 11 mm caused rills to form in areas with convergent flow and a mean sediment yield of 6.2 Mg ha^{-1} (Libohova 2004). These rills rapidly extended to within 10-20 m of the ridgetops, and they continued to incise during each major rainstorm for the first three years after burning (Pietraszek 2006). From 2002 to 2004 the mean sediment yield was 7, 11, and 9 Mg ha^{-1}, respectively.

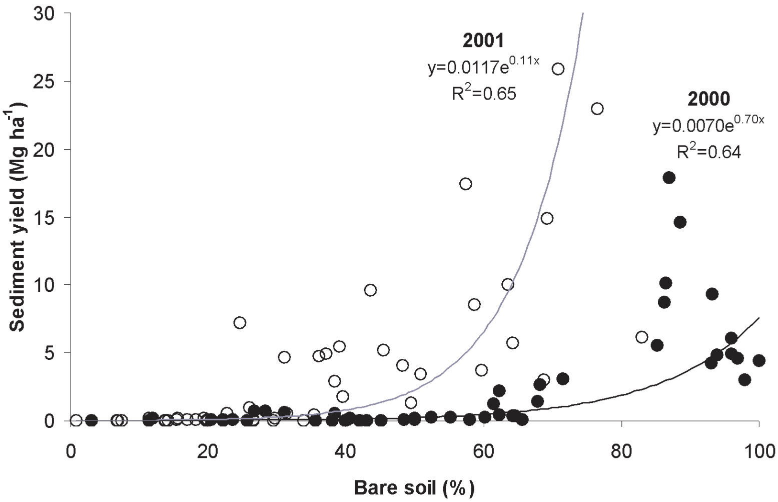

FIGURE 7. Relationship between percent bare soil and annual sediment production rates for 2000 and 2001 from three wild and three prescribed fires in the Colorado Front Range. The greater sediment yields in 2001 were due to a 50-400% increase in rainfall erosivity at each fire.

The importance of topography, concentrated overland flow, and rilling can be shown by the observed differences between planar and convergent hillslopes, respectively. Planar hillslopes on the Bobcat and Hayman fires developed much smaller rills that showed little net incision over time relative to the convergent hillslopes, and unit area sediment yields were three times higher for the convergent hillslopes with central rills than for planar hillslopes (Benavides-Solorio and MacDonald 2005, Pietraszek 2006). Successive measurements of rill cross-sections from the convergent hillslopes showed that rill erosion could account for 60-80% of the sediment collected from the sediment fences (Pietraszek 2006).

In our severely burned hillslopes there was no evidence of sediment deposition, and this was also true for the steep headwater channels below our sediment fences. This means that nearly all of the sediment generated at the hillslope scale is being delivered to the channel network (Pietraszek 2006). Cross-section measurements after the nearby 1996 Buffalo Creek wildfire also showed that channel incision accounted for about 80% of the estimated sediment yield from small catchments (Moody and Martin 2001a). Together these results indicate that rill and channel incision are the dominant sources of post-fire sediment.

Continued monitoring of these and other study sites shows that the median sediment yield from areas burned at high severity decreases by an order of magnitude between the second and third years after burning (Fig. 8), and we attribute this decline to the increase in surface cover as a result of vegetative regrowth. Sediment yields generally return to near-undisturbed levels in 3-5 years in the Colorado Front Range (Fig. 8) (Pietraszek

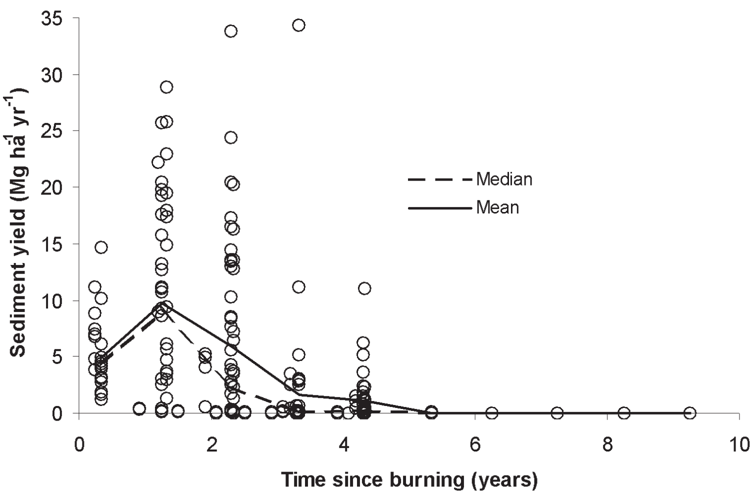

FIGURE 8. Annual sediment yields versus time since burning for six wildfires and three prescribed fires in the Colorado Front Range for high severity burns (from Pietraszek 2006).

2006). A similar recovery period was noted in a seven-year study in a dryland area in Spain, as this showed that catchment-scale runoff and sediment yields were highest in the third year after burning but were very low after five years (Mayor et al. 2007). The long recovery period was attributed to below average rainfall and the correspondingly slow revegetation rate (Mayor et al. 2007). In Colorado we have observed slower vegetative regrowth in areas with coarser soils because of the poorer growing conditions (Pietraszek 2006). The area burned by the 2002 Hayman fire has particularly coarse-textured soils, and after five years the mean amount of surface cover has nearly stabilized at about 65-70%, which means that some sites are still generating some sediment during the larger storm events (MacDonald et al. 2007).

Recent work indicates that the accumulation of sediment in downstream channels may persist for a much longer period than the 3-5 years needed for hillslope erosion rates to recover to pre-fire levels (Eccleston 2008). As noted above, nearly all of the sediment eroded from the convergent hillslopes is delivered to the channel network, but this sediment tends to accumulate in lower-gradient, downstream channels because of the lower transport capacity. In the case of the Hayman wildfire, the first couple of storms caused over 1.2 m of aggradation in some downstream reaches in the 3.4 km^2 Saloon Gulch watershed, and this sediment completely buried an 0.75 m H-flume that had been installed to measure runoff (Libohova 2004). Another 0.2 m of aggradation occurred in this channel over the next four years (Eccleston 2008).

We project that much of the sediment deposited after fires enters into long-term storage, as the combination of vegetative regrowth and the decline in soil water repellency means that hillslope- and catchment-scale runoff rates approach pre-fire values within 3-5 years (e.g., Moody and Martin 2001a, Kunze and Stednick 2006). The decline in runoff means a corresponding decline in sediment transport capacity, and this severely limits the amount of post-fire sediment that can be entrained and transported further downstream. In the nearby Buffalo Creek fire the residence time of fire-related sediment has been estimated to be about 300 years (Moody and Martin 2001a). In other cases, such as the Saloon Gulch watershed, the residence time is likely to be even longer, as in severely aggraded channels most of the runoff is subsurface flow. In watersheds with less aggradation and perennial surface is the channels can more readily return to pre-fire conditions because the streams can slowly excavate the accumulated sediment. In these cases the channels might recover in decades rather than centuries.

The effectiveness of post-fire rehabilitation treatments

After the Bobcat fire, large areas were treated by aerial seeding, while some of the more sensitive areas that burned at high severity were treated with straw mulch at 2.2 Mg ha^{-1} or by contour felling. A 5-10 year storm two months after the Bobcat fire caused three-quarters of the sediment fences to fill with sediment and overflow. Although the sediment fences on the mulched plots were not overtopped, the high erosion rates and high spatial variability meant that none of the treatments had significantly lower sediment yields in the first summer after burning than the controls (Fig. 9) (Wagenbrenner et al. 2006).

In each of the next three years the hillslopes treated with straw much had significantly lower sediment yields than the untreated controls (Fig. 9). The effectiveness of mulching in reducing post-fire sediment yields is attributed to the increase in mean surface cover from

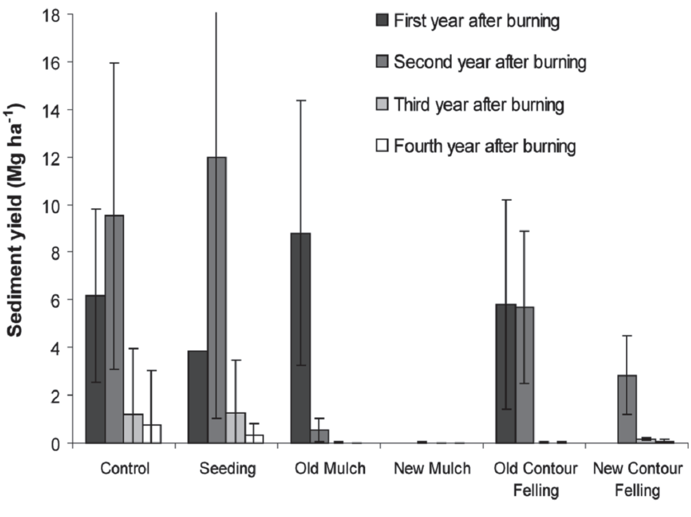

FIGURE 9. Annual sediment yields from treated and control hillslopes at the Bobcat fire. Old much and old contour felling refer to treatments that were applied before a very large storm that occurred two months after the fire. New mulch and new contour felling refer to treatments applied after this storm. Bars indicate one standard deviation.

33% to 75% (Wagenbrenner et al. 2006). In contrast, neither aerial nor hand seeding had any detectable effect on the amount of vegetative regrowth or on hillslope-scale sediment yields (Fig. 9).

The plots treated with contour log erosion barriers prior to the large storm did not significantly reduce sediment yields because the amount of sediment generated by this storm greatly exceeded the sediment storage capacity (Fig. 9). After this storm seven more plots were treated with contour log erosion barriers, and this second contour-felling treatment reduced sediment yields by 71% in the second year after burning ($p<0.05$). In the third and fourth years after burning the sediment yields from these contour-felled plots were 83-91% less than the sediment yields from the adjacent control plots, but this difference was not significant due to high between-plot variability in sediment yields (Wagenbrenner et al. 2006). Detailed surveys of contour log treatments on three fires showed that 32% of the contour-felled logs were completely or partially ineffective in trapping runoff and sediment because they were installed off-contour or had incomplete ground contact (Wagenbrenner et al. 2006). These results indicate that contour felling treatments in the Colorado Front Range are only effective for small- to moderate-sized storms because of the limited storage capacity, and improper installation can be a major problem.

Our studies on the Hayman wildfire generally have confirmed the results from the Bobcat fire. Mulching plus seeding was able to significantly reduce sediment yields relative to the control plots. Seeding plus scarification had no significant effect on the amount of ground cover or sediment yields in any of the first three years after burning (Fig. 10).

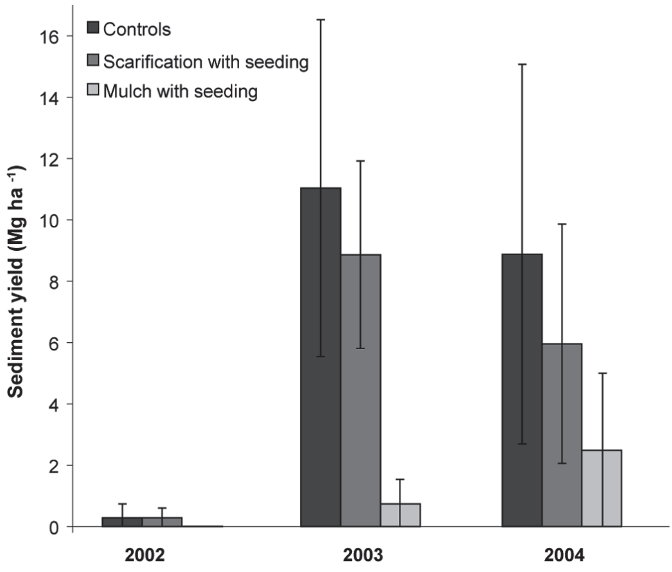

FIGURE 10. Mean annual sediment yields by year from eight untreated (control) hillslopes, four hillslopes treated by scarification and seeding, and four hillslopes treated by mulching and seeding. All sites were burned at high severity by the Hayman fire in the Colorado Front Range. Bars indicate one standard deviation.

Subsequent monitoring has confirmed that seeding and scarification has had no significant effect on either the amount of ground cover or post-fire sediment yields.

Studies in other areas confirm the relative effectiveness of mulching and the general ineffectiveness of seeding in reducing post-fire sediment yields. At the Cerro Grande Fire in New Mexico, the application of straw mulch plus grass seed reduced sediment yields by 70% in the first year after burning and 95% in the second year after burning (Dean 2001). Mulching also reduced sediment yields by an order of magnitude following a wildfire in Spain (Bautista et al. 1996). In contrast, only one of eight studies showed that seeding reduced post-fire erosion (Robichaud et al. 2000). More recently, a four-year study in north-central Washington (USA) showed that neither seeding nor seeding plus fertilization reduced post-fire sediment yields (Robichaud et al. 2006). However, seeding increased surface cover and reduced sediment yields by 550% after an experimental prescribed fire in scrub vegetation in northwest Spain (Pinaya et al. 2000), but it is not clear why seeding was more successful in this particular study.

Comparison of the effects of fires and roads

The sediment production and delivery data from unpaved forest roads and fires allows us to compare the effects of these two disturbances over different time scales at both the hilllslope and watershed scale. Over a five-year period the mean annual sediment production rate from unpaved roads was 42 Mg ha^{-1}, but unpaved roads only occupy about 0.3% of the Upper South Platte River watershed. When the road area is multiplied by the road sediment production rate, the unit area value drops to 0.13 Mg ha^{-1} per year. This converts to 130 Mg ha^{-1} over a 1000-year time span, but the actual sediment production rate over this long time scale would probably be substantially higher because the largest storm events generate a disproportionate amount of sediment (Larson et al. 1997), and the largest rainstorm over the 5-year monitoring period had a recurrence interval of about 6 years. The road connectivity surveys indicate that about 18% of the unpaved roads are connected to the stream network. If all of the sediment from 18% of the roads is assumed to be delivered to the stream network, the watershed-scale sediment yield from unpaved roads over a 1000-year period would be about 23 Mg ha^{-1}. In reality, not all of the sediment from the connected segments would be expected to reach the stream network and be delivered to the South Platte River, but this overestimate is extremely difficult to quantify and may compensate for the likely underestimate of the long-term sediment production rate.

The hillslopes burned at high severity by the Hayman wildfire produced about 10-50 Mg ha^{-1} of sediment before the sediment production rates declined to near-background levels (Pietraszek 2006). The dating of charcoal-rich horizons in alluvial fans at the nearby Buffalo Creek fire indicate that the recurrence interval of large-scale fire and sedimentation events is close to 1000 years (Elliot and Parker 2001). If the erosion rates that we measured after the Hayman fire are assumed to represent one of these millennial scale events, the long-term sediment production from fires is 10-50 Mg ha^{-1} per 1000 years. This value is only about 10-40% of the estimated long-term sediment production rate from roads, but our field observations indicate that nearly all of the sediment from a high-severity fire is delivered to the stream network. If we assume a 100% delivery rate, the long-term sediment yield from fires is 10-50 Mg of sediment per 1000 years. This value is very similar to the estimated sediment delivery rate of 23 Mg ha^{-1} per 1000 years for unpaved forest roads. Again, not all of the sediment will necessarily be delivered to the South Platte River, but nearly all of the stored sediment is potentially accessible for fluvial transport.

The key point is that roads and fires can be expected to deliver a similar amount of sediment to the stream channel network over a 1000-yr period. However, the physical and biological effects of these two sediment sources may be quite different, as the fire-related sediment is being delivered in a large pulse, while the sediment inputs from roads are more continuous. Both fire- and road-derived sediment can degrade aquatic habitat and water quality, and adversely affect algal, macroinvertebrate, and fish populations (Waters 1995). However, native species are generally adapted to the disturbance induced by fires and can quickly recolonize burned areas (Gresswell 1999). The chronic inputs of road sediment do not provide the same opportunities for habitat recovery (Forman and Alexander 1998, Trombulak and Frissell 2000). The implication is that the long-term effects of road erosion on water quality and aquatic ecosystems are at least comparable to, and may be worse than the effects of large, high-severity fires. From a management perspective, the production and delivery of sediment from roads often can be greatly reduced with Best Management Practices, while it is much more difficult to apply mitigation treatments and reduce sediment yields after large, high-severity wildfires. Given the potentially significant effect of road sediment delivery on steams and water quality, forest resource managers should be devoting more effort to minimizing the chronic inputs from unpaved roads rather than trying to reduce the flooding and sedimentation after infrequent, high-severity wildfires.

Conclusions

Undisturbed forests have high infiltration rates and very low surface erosion rates. However, the unpaved roads used to access the forest have low infiltration rates and relatively high surface erosion rates. In drier areas most of the road-related runoff and sediment is unlikely to be delivered to the stream channel network, but as annual precipitation increases road-stream connectivity increases because of the greater travel distance of road runoff and the greater number of road crossings.

High-severity fires are of considerable concern to land managers because they can increase runoff and erosion rates by one or more orders of magnitude. The increases in runoff and erosion are due to the loss of the protective litter layer and subsequent soil sealing, the development of a water repellent layer at or near the soil surface, the disaggregation of soil particles due to the combustion of soil organic matter, and the high runoff velocities due to the loss of surface roughness. After high-severity fires in the Front Range of Colorado, surface runoff is generated by storm intensities of only 7-10 mm h^{-1}. This runoff is rapidly concentrated in topographically convergent areas, and the resultant rill and gully incision is the dominant source of sediment. Sediment yields from areas burned at high severity decline to near-background levels within 3-5 years after burning, and this is primarily attributed to the decline in percent bare soil over time. Runoff and erosion from areas burned at moderate and low severity are of much less concern because these values are commonly 5 or 10 times less than areas burned at high severity.

Rehabilitation treatments that immediately increase the amount of surface cover, such as mulching, significantly reduce post-fire sediment yields. Seeding generally does not increase revegetation rates and therefore is not effective in reducing post-fire sediment yields. Contour-felled log erosion barriers provide a limited amount of sediment storage, so this treatment is only effective in reducing sediment yields from small- to moderate-sized storms.

Over a millennial time scale, the amount of sediment delivered to streams from unpaved forest roads is equal to or greater than the amount of sediment that is delivered from high-severity wildfires. The chronic delivery of sediment from roads may be of greater significance to aquatic ecosystems than the pulsed delivery of sediment from high-severity wildfires, and forest managers should take steps to minimize road runoff and sediment delivery if downstream aquatic resources are being adversely affected.

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