Principles and Practice of Forest Landscape Restoration

Case studies from the drylands of Latin America

Edited by A.C. Newton and N. Tejedor

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This book is dedicated to the memory of Margarito Sánchez Carrada, a student who worked on the research project described in these pages.

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FOREWORD

Every year, a forest area the size of Greece[1] is lost. More than 80% of the world's forests have been cleared, fragmented or degraded. The world's biodiversity and climate, and the livelihoods of hundreds of millions of people are under serious threat.

On behalf of the Global Partnership on Forest Landscape Restoration (GPFLR), the International Union for Conservation of Nature (IUCN), the World Resources Institute and South Dakota State University have begun to map the extraordinary potential of deforested and degraded landscapes for restoration to address the challenges facing societies around the world today and in the future.

This latest research tells us that there are more than 1 billion hectares of lost or degraded forest lands worldwide where restoration opportunities may be found. With this comes substantial potential to not only sequester large volumes of carbon but also to help lift people out of poverty and reduce the vulnerability of rural people and ecosystems through restoration of forest landscapes.

The important role of landscape restoration has been recognized through recent international decisions on climate change and biodiversity. In October 2010, nearly 200 governments attending the Conference of the Parties to the Convention on Biological Diversity in Nagoya, Japan adopted a target calling for restoration of at least 15% of degraded ecosystems by 2020. Just two months later, in December 2010, Parties to the UN Framework Convention on Climate Change, convened in Cancun, Mexico, adopted the goal to slow, halt and reverse forest cover and carbon loss through REDD+ actions.

In February 2011, the UN Forum on Forests called on Member States and others to build on the work of the GPFLR to further develop and implement forest landscape restoration.

If these international commitments are to be translated into action, further evidence of the effectiveness of forest landscape restoration and guidance on how to implement this approach will be needed.

The book *Principles and Practice of Forest Landscape Restoration: Case studies from the drylands of Latin America,* edited by A.C. Newton and N. Tejedor, is an excellent compendium of case studies and analysis, which will be of interest and use to people who wish to move forest landscape restoration forward, no matter what country they operate in.

Practitioners and policy-makers working on forest landscape restoration are learning all the time, through experience, and from each other. It is important to continue to connect partners and collaborators around the world, from Scotland to Sudan and Moldova to Mexico, in a growing community of practice, enabling them to spread best practice, build cooperation and exchange new ideas and solutions.

In this International Year of Forests, this new publication will be an essential contribution to expanding the body of knowledge on forest landscape restoration and strengthening the network of forest landscape restoration experts around the world.

Julia Marton-Lefèvre,
IUCN Director General

[1] FAO, 2005

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ABBREVIATIONS

ACG	Conservation Area of Guanacaste
CBD	Convention on Biological Diversity
CODERS	Consejos de Desarrollo Rural Sustentable
ECOBONA	Programa Regional para la Gestión Social de Ecosistemas Forestales Andinos
EDIEM	Estación de Investigaciones Ecológicas Mediterráneas
ENSO	El Niño Southern Oscillation
FCPF	Forest Carbon Partnership Facility
FLR	Forest Landscape Restoration
FRIS	Forest Restoration Information Service
GAM	Generalised Additive Model
GIS	Geographical Information System
IUCN	International Union for Conservation of Nature
LGDFS	The General Law on Sustainable Forestry Development
LGEEPA	The General Law of Ecological Equilibrium and Environmental Protection
LPI	Large Patch Index
MBA	Man and Biosphere Program
RBYungas	Biosphere Reserve of the Yungas
REDD	Reducing Emissions from Deforestation and Forest Degradation
SL	sustainable livelihoods
SSDF	Subtropical Seasonally Dry Forest
TDF	Tropical dry forests
TEEB	The Economics of Ecosystems and Biodiversity
UNESCO	United Nations Educational, Scientific and Cultural Organization
UNFCCC	United Nations Framework Convention on Climate Change
WWF	Worldwide Fund for Nature (also known as World Wildlife Fund in North America)

1 INTRODUCTION

A.C. Newton and N. Tejedor

The widespread loss and degradation of native forests is now recognized as a major environmental issue. The problem is so acute that it is justifiably referred to as a 'deforestation crisis' (Spilsbury, 2010). Recent reviews indicate that while the rate of deforestation is slowing in some countries, the overall rate of forest loss remains high, estimated at around 130,000 km^2/year during the decade 2000–2010 (FAO, 2010; Secretariat of the Convention on Biological Diversity, 2010). Deforestation figures fail to provide a complete picture, however, as many remaining forests are being severely degraded through the use of fire, cutting and herbivory. Accurate data on the extent of forest degradation at the global scale are difficult to obtain, but an indication of its impact is provided by a recent estimate of the amount of carbon stored in forest vegetation. Over the period 1990–2005, global forest carbon stocks declined by almost double the decline in forest area (UNEP, 2007).

In response to forest loss and degradation, increasing efforts are being directed towards ecological restoration. Forest restoration refers to the process of assisting the recovery of a forest ecosystem that has been degraded, damaged or destroyed (Mansourian, 2005). This may involve the re-establishment of the characteristics of a forest ecosystem, such as composition, structure and function, which were prevalent prior to degradation (Jordan *et al.,* 1987; Hobbs and Norton, 1996; Higgs, 1997). Ecological restoration has been defined in a variety of ways in the past; earlier definitions indicated that the purpose of restoration is the comprehensive re-creation of a specified historical ecosystem, including structural, compositional, and functional aspects. Such definitions emphasize the importance of historical fidelity as an endpoint of restoration. In contrast, more recent definitions allow a more flexible set of objectives, noting that cultural values may be important and that a range of ecological variables can be acceptable as endpoints (Higgs, 1997). Some definitions of concepts relating to forest restoration are presented in **Box 1.1.**

Box 1.1 Definition of some key concepts relating to forest restoration, from Lamb and Gilmour (2003)

Reclamation: recovery of productivity at a degraded site using mostly exotic tree species. Species monocultures are often used. Original biodiversity is recovered but protective function and many of the original ecological services may be re-established.

Rehabilitation: re-establishing the productivity and some, but not necessarily all, of the plant and animal species originally present. For ecological or economic reasons, the new forest may include species not originally present. In time, the original forest's protective function and ecological services may be re-established.

Ecological restoration: re-establishing the structure, productivity and species diversity of the forest originally present. In time, ecological processes and functions will match those of the original forest. The Society for Ecological Restoration defines it as "the process of assisting the recovery of an ecosystem that has been degraded, damaged or destroyed".

Human well-being: Ensuring that all people have a role in shaping decisions that affect their ability to meet their needs, safeguard their livelihoods and realize their full potential.

Definition of Forest Landscape Restoration

Forest Landscape Restoration (FLR) represents a relatively new approach to forest restoration. It was first developed at a workshop led by WWF and IUCN involving a broad range of governmental and non-governmental partners (WWF and IUCN, 2000). This meeting defined FLR as "a planned process that aims to regain ecological integrity and enhance human well-being in deforested or degraded forest landscapes" (Maginnis *et al.*, 2007; Mansourian, 2005).

Since then, the FLR concept has continued to develop. For example:

- Dudley and Aldrich (2007) describe FLR as: "A forward looking approach that aims to strengthen the resilience of forest landscapes and to keep a variety of future options open for benefiting from forests and their products, rather than always aiming to restore forests to their original state";
- Aldrich *et al*. (2004) suggest that: "Forest Landscape Restoration focuses on re-establishing functions and key ecosystem processes across a whole landscape rather than at just planting or restoring individual sites. As such, it looks at a mosaic of land uses including agricultural lands and forest types ranging from plantations to natural forests; aiming for a balanced mixture of protection, management and restoration providing biodiversity, ecological, economic and social benefits and resisting detrimental change";
- Bekele-Tesemma and Ababa (2002) state that: "FLR provides a framework for governments, the private sector, NGOs and conservationists to work with one another in making informed decisions about sustainable land use"; and
- Saint-Laurent (2005) summarizes FLR as "getting the right mixture of approaches, at the right scale, to deliver the forest goods and services that people and societies need".

As noted by Dudley *et al.* (2005), the FLR approach was developed in response to the widespread failure of more traditional approaches to forest restoration, which have often been site-based, focusing on one or a few forest products, relying heavily on tree planting of a limited number of species, and failing to address the root causes of forest loss and degradation. FLR represents a significant departure from such approaches, as presented in **Boxes 1.2 and 1.3**. Further information about the FLR approach, including experience gained in its practical application, is provided by Dudley and Aldrich (2007), ITTO and IUCN (2005), IUCN (2008b), Mansourian *et al.* (2005), Pfund and Stadtmüller (2005) and Rietbergen-McCracken *et al.* (2007).

The development and application of FLR has become a major activity of the WWF and IUCN forest programmes. The approach was further supported by development of the Global Partnership on Forest Landscape Restoration (http://www.ideastransformlandscapes.org/), which now involves more than 25 governments and international and non-governmental organizations, including WWF and IUCN.

Tropical sub-deciduous forest (El Ocote Biosphere Reserve). Photo: R. Vaca

Fires in dry forest areas of Chile. Photo: C. Echeverria

Box 1.2 Elements of FLR, according to Mansourian (2005)

- It is implemented at a **landscape scale** rather than a single site - that is to say, planning for forest restoration is done in the context of other elements: social, economic, and biological, in the landscape. This does not necessarily imply planting trees across an entire landscape but rather strategically locating forests and woodlands in areas that are necessary to achieve an agreed set of functions (e.g. habitat for a specific species, soil stabilization, provision of building materials for local communities).
- It has both a **socioeconomic and an ecological dimension**. People who have a stake in the state of the landscape are more likely to engage positively in its restoration.
- It implies addressing the root **causes of forest loss and degradation**. Restoration can sometimes be achieved simply by removing whatever caused the loss of forest in the first place (such as perverse incentives and grazing animals). This also means that without removing the cause of forest loss and degradation, any restoration effort is likely to be in vain.
- **It opts for a package of solutions**. There is no single restoration technique that can be applied to all situations. In each case a number of elements need to be covered, but how to do that depends on the local conditions. The package may include practical techniques, such as agro forestry, enrichment planting, and natural regeneration at a landscape scale, but also embraces policy analysis, training and research.
- It involves a range of **stakeholders** in planning and decision-making to achieve a solution that is acceptable and therefore sustainable. The decision of what to aim for in the long term when restoring a landscape should ideally be made through a process that includes representatives of different interest groups in the landscape in order to reach, if not a consensus, at least a compromise that is acceptable to all.
- It involves **identifying and negotiating tradeoffs**. In relation to the above point, when a consensus cannot be reached, different interest groups need to negotiate and agree on what may seem like a less than optimal solution if taken from one perspective, but a solution that when taken from the whole group's perspective can be acceptable to all.
- It places the emphasis not only on forest quantity but also on **forest quality**. Decision-makers often think predominantly about the area of trees to be planted when considering restoration, yet often improving the quality of existing forests can yield bigger benefits for a lower cost.
- It aims to **restore a range of forest goods, services, and processes**, rather than forest cover *per se*. It is not just the trees themselves that are important, but often all of the accompanying elements that go with healthy forests, such as nutrient cycling, soil stabilization, medicinal and food plants, forest dwelling animal species, etc. Including the full range of potential benefits in the planning process makes the choice of restoration technique, locations, and tree species much more focussed. It also allows more flexibility for discussions on tradeoffs with different stakeholders, by providing a diversity of values rather than just one or two.
- Forest landscape restoration goes beyond establishing forest cover *per se*. Its aim is to achieve a **landscape containing valuable forests**, for instance partly to provide timber, partly mixed with subsistence crops to raise yields and protect the soils, as well as partly improving biodiversity habitat and increasing the availability of subsistence goods.

Cattle under *Acacia pennatula*, Chiapas, Mexico. Photo: B. Ferguson

Amatenango del Valle, Central Valley of Chiapas, Mexico. Photo: R. Vaca

Box 1.3 Characteristics of FLR approaches, after Maginnis *et al.* (2007)

- **It takes a landscape-level view**. This does not mean that every FLR initiative must be large-scale or expensive; rather that site-level restoration decisions need to accommodate landscape-level objectives and take into account likely landscape-level impacts.
- **It operates on the 'double filter' condition**. Restoration efforts need to result in both improved ecological integrity and enhanced human well-being at the landscape level.
- **It is a collaborative process.** It involves a wide range of stakeholder groups collectively deciding on the most technically appropriate and socioeconomically acceptable options for restoration.
- **It does not necessarily aim to return forest landscapes to their original state.** Rather it is a forward-looking approach that aims to strengthen the resilience of forest landscapes and keep future options open for optimizing the delivery of forest-related goods and services at the landscape level.
- **It can be applied not only to primary forests but also to secondary forests, forest lands and even agricultural land.**

Dryland forests

The problem of environmental degradation is most intense in arid and semi-arid areas (Geist and Lambin, 2004), which together cover nearly 30% of the Earth's surface and comprise half the surface area of the world's developing countries (UNDP, 2004). Despite their aridity, dryland areas are of global importance for biodiversity, being the centres of origin of many agricultural crops and other economically important species. Rural communities in dryland areas are often highly dependent on forest resources to support their livelihoods, particularly fuelwood and fodder. However, in many areas dryland forests have been subjected to unsustainable land-use practices, including expansion of rangeland for livestock, overharvesting (particularly for fuelwood), conversion to agriculture and rapid growth of urban settlements. These processes have resulted in the widespread deforestation and degradation of dryland forest ecosystems, which has resulted in negative impacts on biodiversity, soil fertility and water availability, and on the livelihoods of local people (UNDP, 2004). Such degradation presents a major challenge to policy initiatives aiming to support sustainable development. Restoration of dryland forest ecosystems is therefore an urgent priority if such policy goals are to be achieved, yet this issue has been neglected by the scientific research community.

In 1988, Janzen (1988b) stated that tropical dry forests (TDF) are the most threatened of all major tropical forest types. This statement was largely based on the observation that less than 2% of the tropical dry forest in the Mesoamerican region was sufficiently intact to be considered worthy of conservation, having declined from an original area of 550,000 km^2. At that time, only 0.09% of the forests in the Mesoamerican region were accorded some degree of official protection. While the area under protection has subsequently increased, more recent analyses confirm that remaining dry forests are highly threatened. Miles *et al.* (2006) estimated that 1,048,749 km^2 of tropical dry forest remains throughout the tropics, more than half of which (54.2%) is located in South America. In this region, the two most extensive contiguous areas that remain are located in northeastern Brazil; and southeastern Bolivia, Paraguay and northern Argentina. Other notable concentrations of tropical dry forest occur within the Yucatan peninsula of Mexico, northern Venezuela and Colombia. Overall, approximately 97% of the remaining area of tropical dry forest is at risk from one or more threats including climate change, habitat fragmentation, fire, increasing human population density and conversion to cropland (Miles *et al.*, 2006).

As noted by Miles *et al.* (2006), the definition of dry forest is somewhat problematic, as dry forests grade into other vegetation types such as wet forests, savannahs and woodlands. Mooney *et al.* (1995) suggest that in the simplest terms, tropical dry forest may be defined as forests occurring in tropical regions characterized by pronounced seasonality in rainfall distribution, resulting in several months of drought. The forests that develop under such climatic conditions share a broadly similar structure and physiognomy. However, as noted by Mooney *et al.* (1995), these shared characteristics are difficult to define with precision. Variation in the duration of the rainy season, topography and soil physical characteristics, in particular soil moisture, account for the large differences observed in canopy height, total biomass, productivity and water availability found among these forests (Mooney *et al.*, 1995; Murphy and Lugo, 1995). Dry forests also vary in structure, from relatively open parkland to dense scrub and closed canopy forest (Killeen *et al.*, 1998).

Tropical dry forests have high plant species diversity and endemism (Gentry, 1995; Janzen, 1987; Jansen, 1988b; Kalacska *et al.*, 2004), and although they typically have lower biomass than wet forest, they may be characterized by greater structural and physiological diversity (Mooney *et al.*, 1995). Their growth rate and the regeneration of plants are relatively low and reproduction is highly seasonal. Most plants are out-crossed and dependent on animal pollination (Quesada *et al.*, 2009) and their seed dispersal is principally by wind and animal vectors (Janzen, 1988a).

While the conservation assessment of Miles *et al.* (2006) focused explicitly on tropical dry forests, other dry forest types occur outside the tropics in Mediterranean and temperate climates. In Latin America, these include the Mediterranean forests and shrublands of central Chile, and the temperate dry forests on the eastern side of the Andes in southern Argentina. Both of these areas were also the focus of research described in this book, providing an opportunity to compare and contrast dry forests in different regions of Latin America.

FLR and dry forests

Examples of forest restoration initiatives can be found in the Forest Restoration Information Service (FRIS, 2009), an online database in which more than 200 past and present projects are described. Specific examples of FLR approaches are described by Dudley and Aldrich (2007), Ecott (2002), IUCN (2008a), Mansourian *et al.* (2005), Rietbergen-McCracken *et al.* (2007), and the Global Partnership on Forest Landscape Restoration (2009) (**Box 1.4**). However, relatively few of these examples have been implemented in dryland forest areas; one example is provided by New Caledonia (**Box 1.4**). Many FLR initiatives are still at an early stage of development, and therefore evidence of their effectiveness is typically limited (Aldrich and Sengupta, 2005). The most notable example of a long-term, landscape-scale approach to dry forest restoration is the Area de Conservación Guanacaste in Costa Rica, which has re-established forest over some 70,000 ha of former agricultural land since 1985 (**Box 1.5**). This project provides a powerful demonstration not only of the feasibility of FLR approaches in dry forest, but also the potential benefits that such approaches might provide.

Box 1.4 Examples of FLR initiatives in different parts of the world (Aldrich *et al.*, 2004; Ashmole and Ashmole, 2009; Ecott, 2002; Governments of Brazil and the United Kingdom, 2005; Mansourian *et al.*, 2005)

Brazil: FLR approaches are being applied to the restoration of Brazil's Atlantic forest, which has been highly fragmented and now covers less than 7% of its original area. Restoration efforts are focusing on creation of forest corridors to connect biological reserves.

China: National forest restoration programmes involve 97% of the counties and cities in China. FLR approaches have been implemented in the Minshan conservation area, which covers 33,000 km^2 in Sichuan province.

India: Restoration activities in Gujarat have improved water management, protected forested areas, increased planting of local species, and shifted the economic focus away from timber extraction.

Malaysia: Efforts are being made to restore a forest corridor along the Kinabatangan River, which will connect coastal mangroves with upland forests.

Mali: In the Niger Delta, two forests have been restored, leading to increased fishery production, resolution of conflicts, improved social cohesion and the building of local capacity.

New Caledonia: Less than 2% of the original extent of tropical dry forest remains in this Pacific territory. An action programme has been launched to protect and restore these forests, while contributing to social and economic development.

Tanzania: Since 1985, Sukuma agropastoralists in Shinyanga (northern Tanzania) have restored 250,000 ha of degraded land from semi-desert to forest.

United Kingdom: In the Carrifran Wildwood Project, native woodland has been re-established over an entire sub-catchment of the deforested hills of southern Scotland.

Box 1.5 Area de Conservación Guanacaste (ACG)

The single most significant ecological restoration initiative in neotropical dry forest is the Area de Conservation Guanacaste (ACG) in Costa Rica, an initiative led by Daniel H. Janzen of the University of Pennsylvania. Dry forest has been re-established on approximately 70,000 ha of former fields and pastures since 1985. Key activities undertaken to support this restoration process were the cessation of anthropogenic fires and all harvest of plants and animals; the purchase of enough farm and ranch land to create one large unit of land; and letting the forest recover through natural processes. The initiative also appointed a highly knowledgeable and committed staff, who were instrumental in the success of the project. The ACG represents one of the most significant forest restoration projects ever undertaken in the tropics, not least for its role in inspiring restoration efforts in other areas. The ACG has also played a major role in drawing attention to the conservation value and importance of tropical dry forests, and in demonstrating the feasibility of their restoration. However, the ACG remains one of very few initiatives that have aimed to restore dry forests at the landscape scale. As such, it was one of the main sources of inspiration for the research described in this book. Further information about the ACG is provided by Allen (1988), FRIS (2007) and Janzen (2002).

Research aims and approach

This book presents the results of an international research project, which was designed explicitly to examine application of the FLR approach to dryland forest ecosystems in Latin America. In order for FLR to be transferred into mainstream practice that is adopted and promoted by governments and the private sector, as well as by local communities, information is needed on how the principles of FLR can be implemented in practice, in a cost-effective manner. It is this information need that the project was designed to address.

The ReForLan project (Restoration of Forest Landscapes for Biodiversity Conservation and Rural Development in the Drylands of Latin America; http://reforlan.bournemouth.ac.uk/) was a collaborative research initiative involving ten partners (**Box 1.6**), undertaken during the years 2007–2009 (Newton, 2008). The overall objective of the project was to identify and promote approaches for the sustainable management of dryland forest ecosystems, by researching ecosystem restoration techniques using native species of economic value. This was achieved by undertaking a programme of multi-disciplinary research analyzing how restoration of degraded lands can be achieved in a way that mitigates the effects of unsustainable land-use practices, contributes to conservation of biodiversity and supports the development of rural livelihoods, according to the FLR approach. The project aimed to achieve a strong link between technology, management and policy research.

The specific project aims were to:

- Identify opportunities for enhanced economic productivity and limits to sustainable production, with a particular focus on identifying incentives for supporting dryland forest restoration by local communities;
- Analyze the natural resource use systems at local, regional and international levels through an integrated approach, by developing a comparative programme of research in seven study areas distributed in dryland areas of Mexico, Chile and Argentina;
- Use the information gathered through participatory techniques to inform the planning and implementation of sustainable management strategies for dryland forest resources;
- Develop appropriate decision-support tools, including information systems, criteria and indicators of sustainability and rehabilitation, together with case studies of practical restoration trials, to support dryland ecosystem management and policies; and
- Disseminate the results through scientific publications, research reports and internet resources; strengthen the research capacity of partner organizations both in Europe and Latin America; and provide training and educational resources.

Box 1.6 Principal partners of the ReForLan project

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Coordinator: Professor Adrian Newton

Pontificia Universidad Católica de Chile (PUC), Santiago, Chile
Contact: Prof. Juan J. Armesto

Universidad Austral de Chile (UACH), Facultad de Ciencias Forestales, Chile
Contact: Prof. Antonio Lara

Universidad Nacional del Comahue (UNCO), Laboratorio Ecotono, Bariloche, Argentina
Contact: Dra. Andrea Premoli

Fundación Proyungas (FPY), Tucumán, Argentina Contact: Dr. Lucio R. Malizia

El Colegio de la Frontera Sur (ECOSUR), San Cristobal, Chiapas, Mexico
Contact: Dr. Mario González-Espinosa

Instituto Politécnico Nacional (IPN), Centro Interdisciplinario de Investigacion para el Desarrollo Integral Regional, Oaxaca, Mexico Contact: Dr. Rafael F. del Castillo

Instituto de Ecologia (IE), Xalapa, Veracruz, Mexico
Contact: Dra. Guadalupe Williams-Linera

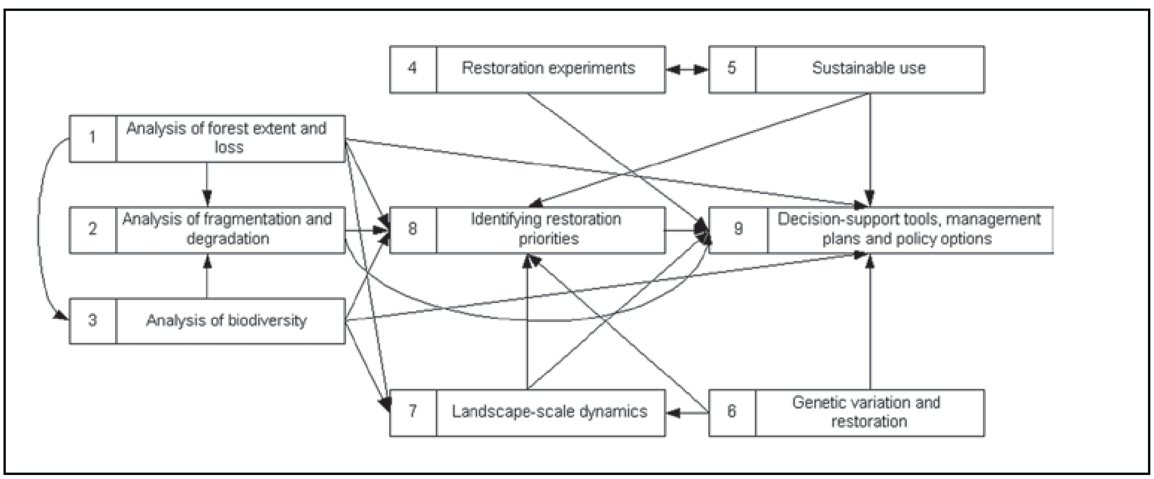
Universidad de Alcala (UAH), Departamento de Ecologia, Madrid, Spain
Contact: Prof. José M. Rey Benayas

Università degli Studi di Trento (UNITN), Dipartimento di Ingegneria Civile e Ambientale, Università degli Studi di Trento, Trento, Italy Contact: Dr. Davide Geneletti

Additional partners included the UNEP World Conservation Monitoring Centre, Cambridge, UK, (Contact: Dr. Lera Miles), Universidad de Concepción, Concepción, Chile (Contact: Dr. Cristian Echeverría), Universidad Veracruzana, Xalapa, Mexico (Contact: Dra. Claudia Alvarez Aquino).

The research was implemented as a series of nine interconnected Work Packages (**Fig. 1.1**), which form the basis of the structure of this book. This book profiles the results of this research, but also presents some additional results provided by other related projects in the region, which were invited contributions to this volume.

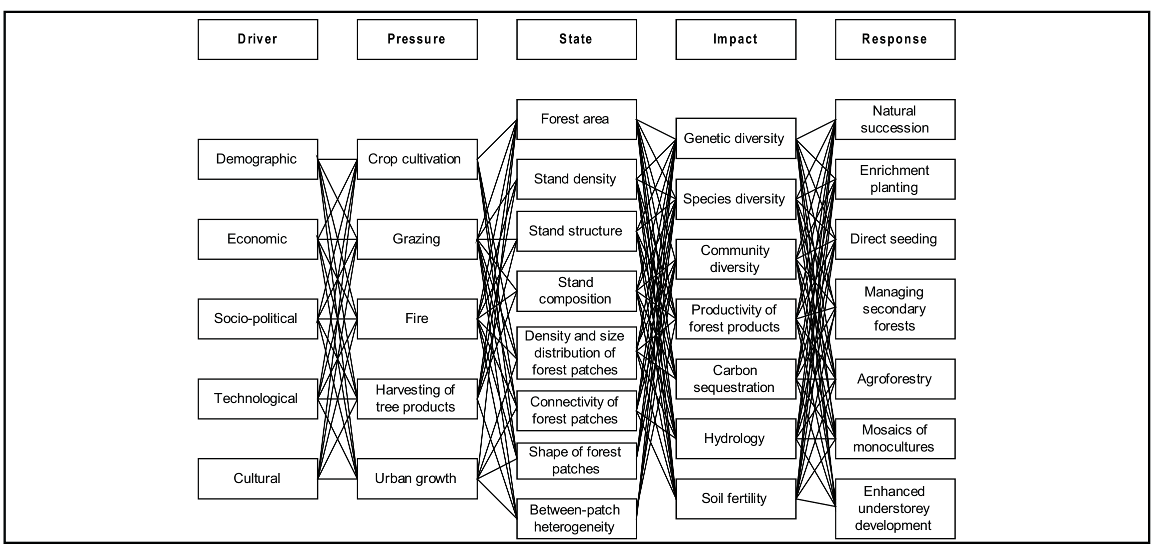
Figure 1.1 Interrelationships of different elements (Work Packages) of the ReForLan project.



Overall, the research aimed to identify how dryland forest ecosystems could be restored in ways that both benefit biodiversity and support the livelihood of local communities, and thereby contribute to sustainable development objectives. A conceptual framework was developed at the outset of the project to provide a basis for organizing and integrating research activities (Newton, 2008). This was based on the consideration of forest restoration as a potential response to environmental degradation caused by unsustainable land-use practices. Such response options can usefully be viewed according to the DPSIR (Driving forces – Pressures – State – Impacts – Responses) framework, which was developed by the European Environment Agency to help analyze the process of sustainable development (EEA, 1998). The DPSIR framework is based on the fact that different societal activities (drivers) cause a pressure on the environment, which can cause quantitative and qualitative changes in the state of environmental variables. Such changes can produce a variety of different impacts on natural resources and the services that they provide to human communities. Society has to respond to these changes in appropriate ways in order to achieve sustainable development. According to the DPSIR framework, different indicators of sustainability can be developed relating to driver, pressure, state, impact and response variables; the development of such indicators was one of the outputs of the project.

The research approach was based on the application of the DPSIR framework to restoration of dryland forest resources (**Fig. 1.2**). The underlying drivers responsible for unsustainable land-use patterns can be grouped into demographic, economic, sociopolitical, technological and cultural factors (Geist *et al.*, 2006; Geist and Lambin, 2001). For example, key factors underpinning current patterns of land-use and land-cover change in dryland regions of Latin America include the current policy context, the structure and function of national and international market chains for agricultural and forest products, and the process of globalization. Such factors influence patterns of land use, such as cultivation of crops and animal husbandry, which can have a major effect on the extent and condition of forest resources. Key variables describing the state of forest resources include forest area, the size distribution and connectivity among forest patches, and the composition and structure of forest stands (**Fig. 1.2**). The way that human activities influence these patterns will determine their impact on key ecological processes, such as dispersal, growth, survival, competition, succession and gene flow, which affect biodiversity and the provision of the environmental services on which human communities depend (**Fig. 1.2**). The severity and extent of environmental degradation, and its impact on biodiversity and the provision of environmental services, will determine both the need and scope for forest restoration as a response option.

Figure 1.2 Schematic diagram illustrating the context of forest restoration as a response to unsustainable land-use practices, according to a DPSIR framework (see text).



Study areas

The research focused on seven dryland areas where native forests have been subjected to intense human pressure in recent decades, resulting in severe deforestation and degradation. Each of these areas is characterized by high biodiversity of international conservation importance, with many endemic, threatened species. These areas are also characterized by the presence of substantial and increasing rural populations, often including indigenous communities, who rely on native forest resources for provision of a number of forest products. The sustainable management of forest resources in these areas is therefore of key importance to the livelihood of local communities. Although the processes of forest degradation in these areas are similar, the socioeconomic and policy contexts vary, providing scope for comparative analysis.

Figure 1.3 Location of the study areas included in the ReForLan project.

The areas where research was primarily conducted are (**Table 1.1**):

- *Central Chile, including the Central Valley and Coastal Range,* extends over 13,175 km^2, covering parts of the Valparaíso, Libertador, Bernando, O'Higgins, and Metropolitan administrative regions. This area lies within the Mediterranean bioclimatic zone, characterized by dry summers and wet winters with strong inter-annual variability. Mean annual temperature is 13.2°C and mean annual precipitation is 531 mm, with elevations ranging from sea level to 2260 m a.s.l. A spatially heterogeneous mosaic of shrublands and evergreen sclerophyllous forests is distributed mostly on the slopes and in drainage corridors. The region is acknowledged as one of the world's 25 biodiversity hotspots (Myers *et al*., 2000). Logging and land-use changes have led to profound landscape transformations since the mid-sixteenth century and agriculture is mostly concentrated in flat valleys, where major activities include vineyard and fruit cultivation as well as corn and wheat cropping. Forest resources are mainly used for the extraction of fuelwood from native tree and shrub species, and extensive livestock husbandry in shrubland and forest areas. Timber plantations are located mainly in the flat coastal zone. The study area is home to around 5.2 million inhabitants, which represents ca. 34% of the Chilean population.
- *Southern Argentina,* at the forest-steppe ecotone in northwestern Patagonia, Argentina. A rectangular study area was delimited with corners: NW: 38° 44', 71° 25'; NE: 38° 46', 70° 48'; SW: 43° 05', 71° 35'; SE: 43° 05', 70° 55'. Total area encompassed is 2.8x10^6 ha; after classification, total non-grassland area was ca. 0.93x10^6 ha. Overlaying the glacial topography, numerous layers of volcanic ash along with sediments from volcanic and pyroclastic rocks constitute soil parent material. Dominant soils are acid and alofanic (Andisols). Owing to the rain shadow effect of the Andes on the westerly winds, mean annual precipitation declines from 3000 mm at the continental divide to 400 mm over only 100 km, in the eastern foothills. Approximately 60% of the annual precipitation falls in the winter season (from May to August). Only 5% of the Argentinean population lives in Patagonia and it is one of the least densely populated areas in the world with 1.8–5 inhabitants/km^2 (INDEC national statistics, Census 2001). In the first decades of the twentieth century, the regional economy was based almost entirely on the sheep industry, which expanded vigorously. The physiographic characteristics of the area have made it an ideal location for ranching. However, the range of economic activities has increased in the past couple of decades, leading to a shift from sheep ranching towards cattle ranching. Forests, lakes, plentiful rivers, and vast grazing areas provide a unique assortment of livelihood opportunities for the rural population.
- *Northern Argentina,* research was carried out in a forest area of ca. 800,000 ha, which corresponds to seasonally dry premontane forests (the lower and drier end of Yungas forest) and the transition to the drier Chaco forest. This area is located in the San Martín department of Salta province. The northwestern extreme of this area is an international border with Bolivia, the southern part borders the Bermejo River and the eastern part reaches the Yungas limit according to Brown and Pacheco (2006). Rainfall ranges from around 1000 mm per year on the western side to 700 mm per year on the eastern side, with a marked seasonal variation determined by rainfall concentration during the summer months (November-March). Maximum temperatures occur in this season, which may exceed 40ºC (Brown and Kapelle, 2001). According to the national statistics (INDEC), San Martin department has 139,204 inhabitants, with a population density of 8.6 inhabitants/km^2 (Census 2002). The study area occupies almost 50% of this department, and includes nearly all

of the most important cities of the area. The forested areas are critical for maintenance of regional biodiversity, environmental services (such as irrigation and land stability), and the sustainable development of the forestry sector. These forests are also important for rural and suburban communities since they provide income through wood, hunting, medicinal plants, honey, and other products. However, in recent decades, deforestation for agricultural purposes has been the main activity associated with land-use change in this area, consistent with the rest of the premontane forest areas and its transition to dry Chaco forest.

- *Central Veracruz, Mexico*, covers a total area of 160,699 ha and is located along the coastal plain from 0 to 800 m. The boundaries (upper left corner: 19°21'20" and 96°50'52.36"; lower right corner: 19°05'51.43" and 96°06'31.37") were determined using the following criteria: (1) the area should be centred around the municipality of Paso de Ovejas; (2) it should include the major sub-watersheds (five in all; CONABIO, 1998) comprising the northern and southern limits of the study area; and (3) the coast of the Gulf of Mexico and 800 m contours correspond to the altitudinal limits of tropical deciduous dry forest in the region (Rzedowski, 2006). Climate is defined as warm sub-humid (mean minimum and maximum average temperatures are 14 to 36°C, respectively), with rainfall (800–1200 mm) occurring primarily in the summer (95%) and followed by an extended (4–5 months) dry season (García, 1990). Topography is very heterogeneous with steep canyons marking the limits of 119 micro-watersheds. The coastal zone is formed mainly by littoral material deposited by wind, except in the centre, which is dominated by sandstone and conglomerate soils. Soil types are very diverse but dominated by the feozem haplic, litosol, and vertisol pelico varieties. All or part of 12 municipalities, and 490 (<1000 inhabitants) and 15 urban localities are located within the study region. While private lands dominate the region, the 151 communal properties ('*ejidos*') are also important, occupying 41% of the study area.

- *Central Valley of Chiapas, Mexico*, also known as Central Depression of Chiapas, is located in the central portion of the state. The boundaries of the Depression dry forest ecotone are: 17°39'28"N and 14°32'00"S; 90°22'28"E; and 94°14'13"W extending over 13,974 km^2 (Olson *et al.,* 2001). This dry valley is more than 200 km long and up to 70 km broad. The strata consist of mostly marine limestones and slates (Breedlove, 1981; Challenger, 1998). Dominant soils are luvisols and lithosols, rendzinas in upper sites (SEMARNAT, 1998). The most influential factor on the climate of this region is the topography. The northern and central highlands of Chiapas protect the valley from the effects of the '*alisios*' and '*nortes*' winds from the Gulf of Mexico. The Sierra Madre of Chiapas produces a rain shadow effect, reducing rainfall as a result of humid winds from the Pacific Ocean. The region therefore has a summer precipitation regime of convective character (Challenger, 1998). According to the WorldClim database, the mean annual temperature of the ecoregion ranges from 22 to 25.2°C, and annual precipitation ranges from 750 to 1500 mm. Most of the valley was probably originally covered with tropical deciduous forest, however extensive cultivation and grazing have led to large tracts of thorn woodland and savannah (Breedlove, 1981; Challenger, 1998). The dry forest of the Central Valley is completely surrounded by moist forested mountain areas, resulting in their relative isolation from other areas of xeric vegetation. The flora contains a number of central Mexican elements, but appears to lack many species commonly found among the dry flora of Oaxaca. Another interesting facet of the flora of this region is the occurrence of many species only found in the dry regions of the Yucatan Peninsula (Breedlove, 1981). Tuxtla Gutierrez, the capital city of the state, is located in the study area and is the largest urban centre in the south of Mexico, with a population of 490,455 inhabitants. Rural population density varies over the area, but is generally below 20 inhabitants/km². This is comparatively low for the state of Chiapas.

- *Oaxaca, Mexico,* research focused on the Mixteca Alta, which belongs to the mountains and valleys of the western Oaxaca physiographic region. This region is located north-west of Oaxaca central valley, and is characterized by a complex topography of canyons, plateaus, valleys and mountains, with creeks and rivers that drain to the Pacific Ocean through the Balsas River. An area of 11,637 km² was surveyed. We limited our study area by including all the municipalities with at least 90% of their land area located within the Mixteca Alta. Elevation ranges between 700–3200 m. Rainfall is concentrated during the summer months, mostly in short and intense showers facilitating soil erosion, with a dry season extending from November to April. Mean annual precipitation is 692 mm and the mean annual temperature is 22°C. Superficial geology is complex: Precambric gneiss, Paleozoic schist, Jurassic sandstone and shale, and Cretaceous limestone, limonite, and sandstone. Vegetation types are variable: mixed pine-oak, pine forests, tropical dry forest and shrublands (CONABIO, 1998). Handicraft manufacturing and agriculture (corn, bean and wheat) are the major economic activities. Croplands are mostly concentrated in the valleys, and to a lesser extent on the mountains. The total population in the study area is 340,000 inhabitants (9.8 % of the state population).

Grazing in the drylands of central Chile. Photo: J. Birch

Table 1.1 Characteristics of the study areas that featured in the ReForLan project.

Country	Study areas	Location: lat. & long.	Elevation (m)	Rainfall (mm)	Soil / geology	Forest type	Land use	Land ownership	Forest characteristics	Primary threats
Mexico	Chiapas	14°32'00" and 90°22'28"; 17°39'28" and 94°14'13"	350–1200	750–1500	Luvisols and lithosols, rendzinas in upper sites	Dry tropical	Agriculture, cattle ranching, forest extraction	Ranches, *ejidos* communal lands	Secondary disturbed forest. Remnant degraded forest on slopes.	Agriculture, cattle, fire
	Central Veracruz. Acazonica, Paso de Ovejas, Puente Naciona, Emiliano Zapata.	19°21'20" and 96°50'52.36"; 19°17'06.36" and 96°26'27.10"	40–800	800–1200	Mainly cambisols and vertisols with considerable exposed rock.	Dry tropical	Livestock husbandry, rain-fed agriculture, irrigated agriculture, fruit-tree plantations	Private land, *ejidos*	Fragmented within a landscape composed of secondary forest, agriculture and grasslands. One-third of the study area is classified as disturbed secondary forest (22%) and forest (9.26%).	Cattle, agriculture
	Oaxaca. Mixteca Oaxaqueña, Central Valley, Sierra Madre de Oaxaca	18°13'28"N –98°20'36"W to 18°12'30"N, –96°25'16"W and 16°44'10"N, –96°26'43"W to 16°45'15"N, –98°20'55"W.	1400–3120	500–1000	Lithosol, vertisol, acrisol and exposed rock.	Dry tropical	Agriculture, forestry, cattle, sheep and goat ranching	Mostly communal	Tropical deciduous forest, oak forest, pine forest, grassland, and shrublands. Small forest remnants on knolls; one of the most extreme cases of environmental degradation in Mexico. High soil erosion.	Agriculture, forestry, livestock, soil erosion
Chile	Central Valley	33° 58 S, 70° 58'W	200–1200	700–1000	Cretaceous and Tertiary formations, mainly granodiorite, tonelite and adamelite	Sclerophyllous, Deciduous dry	Agriculture, fuelwood harvesting, livestock husbandry, mining	Private land principally	Endemic Chilean sclerophyllous dry forests; one of the five Mediterranean climate forests in the world. High endemism, highly degraded soil.	Fuelwood harvest, livestock, urban expansion, agriculture

Table 1.1 Characteristics of the study areas that featured in the ReForLan project (cont.).

Country	Study areas	Location: lat. & long.	Elevation (m)	Rainfall (mm)	Soil / geology	Forest type	Land use	Land ownership	Forest characteristics	Primary threats
	Coastal Range	33° S, 71° 30'W	0–1900	200–450	Alfisol and inceptisols derived from granitic rock and marine terraces respectively	Sclerophyllous and xerophytic plant species such as *Cactus* spp.	Agriculture, pasture, mining, commercial forest plantations	Private and public land	Hard-leaved species; pseudo-savannah. High endemism, highly degraded soil.	Agriculture, exotic tree plantations, land-cover change, mining, soil erosion, urban expansion
Argentina	Northwest Salta & Jujuy Provinces	22°–24°S, 63.5°–65°W	350–750	500–900	Sierras subandinas	Subtropical seasonal dry Andean premontane (Yungas) Dry Chaco	Agriculture, logging, extensive cattle ranching	Private land; has the highest concentration of ethnic groups (9) in Argentina.	Highly seasonal. The Chaco is the largest remaining area of neotropical dry forest and harbours the highest proportion of deciduous species in South America. Forest loss to agricultural land has created an agricultural gap 5–25 km wide between dry Chaco and Andean premontane forest.	Agriculture, land-cover change, selective logging, fire
	Southwest, northern Patagonia	39°30' and 43°35' S	350–1300	600–1000	Poorly developed Andisols from volcanic ash	Forest-steppe ecotone monospecific conifer forest	Sheep, cattle and goat ranching, exotic plantations	Private land	Isolated conifer forest patches. Remnant forest on slopes and in ravines. High fire frequency	Cattle, sheep, goats, exotic tree plantations, fire

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2 ASSESSING THE CURRENT EXTENT AND RECENT LOSS OF DRYLAND FOREST ECOSYSTEMS

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Introduction

Land-cover change is regarded as the most important global change affecting ecological systems (Vitousek, 1994). Natural landscapes – i.e. those largely unaffected or hardly affected by human activities – are being rapidly transformed into urban and farmland landscapes throughout the world (Foley *et al.*, 2005; Feranec *et al.*, 2010; López and Sierra, 2010). As the characteristics of land-cover have important impacts on climate, biogeochemistry, hydrology, species diversity, and the well-being of human societies, land-cover change has been identified as a high priority for research and to inform the development of strategies for sustainable management (Turner *et al.*, 1993; Ojima *et al.*, 1994; Millennium Ecosystem Assessment, 2005a). In recent years, special attention has been given to land-use changes and degradation in drylands. Dryland forests are highly prone to degradation and desertification on account of their limited primary productivity and slow recovery following human disturbance (Millennium Ecosystem Assessment, 2005b), yet these ecosystems play a crucial role in providing services such as climate and water regulation (Maass *et al.*, 2005; Lemons, 2006).

To develop approaches for the conservation and restoration of dryland forests at the regional level it is crucial to know their current extent and to understand the main recent and historical changes that have affected them (Schulz *et al.*, 2010). To accomplish this goal, it is necessary to assess what processes may be driving such changes, to reveal the threats to forest ecosystems, and to develop alternative strategies to diminish these threats (Angelsen and Kaimowitz, 1999; Geist and Lambin, 2002; Lambin *et al.*, 2003; Antrop, 2005; Olander *et al.*, 2008). Changes in patterns of forest distribution at a wide variety of spatial scales, from global to local scales, are among the land-cover changes most frequently investigated. At the global scale, forest extent is declining mostly as a result of expansion of farmland (Foley, 2005; FAO, 2010). Additional activities associated with widespread deforestation and forest degradation include industrial tree plantations (often composed of non-native species), logging for firewood, and cattle grazing (Lara and Veblen, 1993; Kahn and McDonald, 1997; Geist and Lambin, 2002). However, at local and regional scales, land abandonment resulting mainly from rural-urban migration can contribute towards passively restoring considerable amounts of the original forest extent (Aide and Grau, 2004; Pascarella *et al.*, 2000; Rudel *et al.*, 2005; Grau and Aide, 2008; Parés-Ramos *et al.*, 2008). These landscape processes have rarely been mapped and quantified, and change trajectories among land-cover types have not been systematically examined for particular types of forest.

We addressed these issues in selected areas in Chile, Argentina and Mexico, using standardized research protocols (**Box 2.1**). The primary advantage of assessing these different areas is that they include a range of ecological, socioeconomic, and cultural characteristics. We used remote sensing data to measure and monitor land-cover change because of their ability to capture an instantaneous synoptic view of a large part of the Earth's surface and to provide repeated measurements of the same area on a regular basis (Donoghue, 2002). Land-cover change detection and monitoring is especially useful in those regions where there is a lack of available cartographic information with sufficient spatial resolution to examine land-cover change. To investigate the possible causes of change in forest cover, a Geographic Information System (GIS) database incorporating satellite imagery and biophysical and socioeconomic variables was developed for each study area. This information was statistically analyzed to infer likely drivers of forest-cover change and to test a series of specific hypotheses relating to the factors responsible for deforestation. For example, we expected that the rate of forest loss would be (i) positively associated with population density and accessibility (i.e. proximity to roads and rivers), and (ii) highest on sites most suitable for agriculture such as those with gentle slopes (Sader and Joyce, 1988; Pfaff, 1999; Lambin *et al.*, 2003).

As part of the contribution to the international ReForLan project (Newton, 2008), we report here the first multi-regional assessment of land-use/land-cover changes, with special attention to forest loss, spanning a ca. 30-year period (1970s–2000s) in dryland Latin American regions. The results presented should be useful for planning the restoration and conservation of dryland forest in the study areas and elsewhere in the region. Furthermore, they provide an example of the research that needs to be conducted in other regions of the world.

Box 2.1 Methodology used to assess the amount and the drivers of forest change

A total of six areas were considered in this study in central Chile (the Central Valley and the Coastal Range extending to the Pacific Ocean), southern Argentina (northwestern Patagonia), northern Argentina (Salta province), as well as central Veracruz (central Mexico by the Gulf Coast), Central Depression of Chiapas, and Oaxaca (southern Mexico by the Pacific Coast).

Analysis of land-cover/land-use change

Remote sensing data

We acquired a time-series of satellite imagery to analyze land-cover/land-use change in each study area (**Table 1**). All images were pre-processed, including geometric, atmospheric and topographic corrections. The images were geometrically corrected using standard procedures based on ground and roadway map control points. For the removal of atmospheric effects and variations in solar irradiance, an atmospheric correction was carried out to transform the original radiance images to reflectance images using an algorithm based on the Chavez reflectivity model (Chávez, 1996) in most study areas. Topographic corrections were also performed to reduce shadows on hilly areas when necessary. We employed a variety of methods for this task, such as the C-correction proposed by Teillet *et al.*, (1982) using a digital elevation model (DEM) interpolated from contour lines of 25 m for TM and ETM+ images in central Chile and Oaxaca, and the NASA SRTM model with a resolution of three arc seconds per pixel (ca. 80 m) in central Veracruz. To compare images of different pixel size, the original MSS raster grids were re-sampled to the resolution of the TM raster grids (30 m) in most study areas.

Box 2.1 (cont.)

Table 1 Time series of satellite imagery (Landsat and SPOT) used to analyze land-cover/land-use change in the selected study areas in Latin America.

Study area	Year/Sensor	Year/Sensor	Year/Sensor	Year/Sensor
Central Chile	1975 MSS	1985 TM	1999 TM	2008 ETM+
Southern Argentina	1973 MSS	1985 TM	1997 TM	2003 ETM+
Northern Argentina	1976/77 MSS	1987 TM	1993 TM	2006 TM
Central Veracruz (Mx)	1973 MSS	1990 TM	2000 ETM+	2007/08 SPOT
Chiapas (Mx)	—*	1990 TM	2000 ETM+	2005 ETM+
Oaxaca (Mx)	1979 MSS	1989 TM	2000 ETM+	2005 SPOT

*Classification results for 1975 were excluded due to inconsistencies related to Landsat MSS resolution.

Land-cover/land-use classification

All study areas attempted to follow a common protocol for classification of land-cover/land-use types that distinguished eight major classes: (1) forest, (2) shrubland, (3) pasture, (4) bare ground, (5) agriculture, (6) timber plantations, (7) urban areas, and (8) water. However, these pre-defined classes were modified according to local conditions in each study area. The land-cover maps were derived using a supervised classification procedure in all study areas except in northern Argentina, where an ISODATA unsupervised classification technique was used.

To classify the images, field points were taken with a GPS in order to train the spectral signature of the selected land-cover classes (198 in central Chile, 311 in southern Argentina, 1071 in central Veracruz, and 50 in Oaxaca). This information was complemented with high-resolution imagery obtained from Google Earth and aerial ortho-photos to account for areas with restricted accessibility, control points from vegetation and land-use maps. In addition, informal interviews were conducted with land owners and land managers during field surveys to obtain information on previous and current land-cover and land use.

For central Chile and Oaxaca, a region-growing approach was used with the 'seed' function, and signature separability of the initial classes for all images was evaluated using the Bhattacharyya distance. Based on this distance, classes were iteratively merged until reasonably high signature separability (Bhattacharyya distance >1.9) was achieved. For northern Argentina, the classifications performed used nine iterations and a convergence of 0.95. For central Veracruz, polygons were created by means of a regional growth algorithm that combined pixels that were judged to be similar based on the values of the spectral bands available for each image, as well as data from scale, texture, and shape indices. For central Chiapas, classifications were performed using an iterative procedure developed for classifying multi-temporal Landsat imagery in complex tropical landscapes (Harper *et al*., 2007). A unified classification for the three analyzed years was produced. Maximum likelihood classification was the principal method used as it has proven to be a robust and consistent classifier for multi-date classifications (Yuan *et al*., 2005); however, the Sequential Maximum A-Posteriori classifier (SMAP in GRASS 6.4) produced the best results in Chiapas. Post-classification processing was applied to combine initial classes and to better discriminate between confounding classes in each study area. More details on the classification procedures and used software can be found in Rey Benayas *et al.,* (2010a), Schulz *et al.,* (2010), Cristóbal *et al.,* (in prep.), Gowda *et al.,* (in prep.), Manson *et al.,* (in prep.), Rivera *et al.,* (in prep.), and Vaca *et al.,* (in prep.).

Box 2.1 (cont.)

Accuracy assessment

Accuracy assessment of the classification results was carried out using independent ground control points sampled in the field (280 in central Chile, 432–520 control points in southern Argentina, 157 in northern Argentina, and 300 in Oaxaca) as well as control points derived from Google Earth and ortho-photos (>400 in central Veracruz, and >2500 in Chiapas). Based on these points, confusion matrices and associated Kappa indexes of agreement for each class were generated (Rosenfield and Fitzpatrick-Lins, 1986). As independent data sources such as previous classifications and ortho-photos were largely absent for central Veracruz, an Index of Ecological Congruence (IEC) was developed following the same logic as the Kappa index, where congruent and incongruent events were considered as feasible or non-feasible land-cover transitions between two time periods given current ecological understanding of land-use patterns and rates of vegetation succession in the region. Overall classification accuracies for the different classified study areas are reported in **Table 2**.

Table 2 Classification accuracy (%) in the different study areas and images. The sequence of images (columns) follows **Table 1**.

Study area	Series 1	Series 2	Series 3	Series 4
Central Chile	68.5	77.3	78.9	89.8
Southern Argentina	69.3	80.7	85.0	86.0
Northern Argentina	68.2	82.5	85.3	83.4
Central Veracruz (Mx)*	93.4	94.0	74.8	82.0
Chiapas (Mx)	—	74.1**		
Oaxaca (Mx)	92	90	93.3	93.5

*Based on an Index of Ecological Congruency (IEC) and not Kappa for Series 1–3 (1973–2000). **Since a unified classification for the three analyzed years was produced, we performed a unique validation analysis.

Change identification

The spatial distribution of land-cover/land-use changes was investigated using the previously classified remotely sensed images in order to obtain a matrix of change directions among land-cover classes (Lu *et al.*, 2004) in each study area. Changes were analyzed by cross-tabulation as proposed by Pontius *et al.,* (2004) to quantify net changes, gains, losses and persistence as well as inter-categorical change trajectories. In Chiapas, the classification was combined with results from the map series IV (2007–9) of Use of the land and Vegetation elaborated by INEGI (scale 1:250,000).

Drivers of forest cover change

GIS analysis and explanatory variables

To analyze the drivers of forest cover change we created binary maps to represent forest versus other cover types. The points where change (deforestation) occurred or not were determined by overlapping the binary land-cover maps from the two dates that spanned the time period studied in each study area. In southern Argentina, since afforestation and not deforestation was detected, we analyzed drivers of afforestation with exotic conifers rather than drivers of deforestation.

Box 2.1 (cont.)

We then followed a common protocol (Echeverría *et al*., 2006) that included randomly selecting a grid of sampling points separated by a minimum distance of 1000 m for extracting values of both the response (change vs. no change) and explanatory variables (biophysical and socioeconomic) in order to reduce problems with spatial auto-correlation. A number of biophysical and socioeconomic variables that may influence forest cover change were selected for analysis including: (1) elevation (m), (2) slope (°), (3) insolation or radiation input, (4) mean annual precipitation (mm), (5) soil quality, (6) distance from rivers and lakes (m), (7) distance from forest edges (only for forest loss), (8) distance from forest (only for forest gain in southern Argentina), (9) distance from human settlements (cities, towns and villages) with different numbers of inhabitants (m), (10) human population density (#/km^2), (11) distance from different types of roads (e.g. primary or secondary, paved or unpaved (m), (12) distance from the agricultural frontier (m), (13) distance from irrigation infrastructure (m), and (14) distance from cattle pastures (m). For all explanatory variables, values were extracted using the random sampling points previously selected with forest to non-forest changes for each of the four analysis periods. However, not all of these variables were used in each study area as availability of information depended on local conditions.

Model building

Logistic regressions were performed to explore the effect of the described explanatory variables on deforestation in all study areas except in Chiapas, where a different method was used (see below). To fit the models, we started by using the full set of explanatory variables described above. Before starting the model selection process, a Pearson's or Spearman's correlation test was performed to identify the correlated explanatory variables, and the single representative of variables that were highly correlated (typically $r > 0.7$) was selected for further analyses to avoid multicollinearity. A spatial correlogram based on Moran's index of autocorrelation was used to explore the autocorrelation of data at different geographic distances, and was found to be low in all cases. In southern Argentina, the effect of each predictor variable on the occurrence of afforestation was evaluated by univariate logistic models. In Chiapas, a series of generalized additive models (GAMs) of the binomial family were fitted using the R package 'MGCV' (Wood, 2004). These models allow non-linear responses to be modelled.

Model selection

Multivariate, spatially explicit models were developed for each of the three periods of time and the whole study period in most study areas. In central Chile and northern Argentina, we performed a backward stepwise model selection based on the Akaike (1974) Information Criterion (AIC) to determine the set of explanatory variables constituting the best fitting model for each period. The measure used by generalized linear models, including logistic regression, to assess the aptness of fit is called the deviance. Deviance reduction (D^2) is estimated as D^2 = (Null deviance – Residual deviance)/Null deviance. For southern Argentina, selection of the final multivariate logistic model to explain afforestation was performed using forward, backward and best subset procedures with GIS Landchange Modeler (Idrisi, 2006); then a spatial model of potential afforestation was generated and evaluated by means of the relative operating characteristic (ROC) curve. For central Veracruz, those variables with $p \leq 0.10$ in univariate logistic regressions were used to construct a final multiple logistic regression model that best explained the loss of undisturbed forest in the study region. In Chiapas, to complement the GAM analysis and provide a direct interpretation of the strength of the drivers, recursive partitioning was also used as implemented in the R package 'rpart' (Therneau and Atkinson, 2009). Recursive partitioning models allow interactive effects between variables to be investigated.

Major changes in land-cover as a result of land-use intensification

Important changes in land-cover/land-use types occurred during the time period analyzed within each study area, as shown by the mapping and quantification of remote sensing imagery (**Fig. 2.1**) and the analysis of change trajectories among land-cover types (**Fig. 2.2**). The observed changes between land-cover classes differed significantly among the various study areas and between the particular periods of time under study. We detected the following major trends: (1) forest degradation to shrubland (in central Chile and southern Argentina) or secondary forest (in northern Argentina and Veracruz), (2) conversion of shrubland and secondary forest to agricultural land, grassland or bare ground (in central Chile, Veracruz and Chiapas), (3) direct conversion of forest to agricultural land or grassland (in northern Argentina and Chiapas), (4) conversion during different periods between forest and shrubland and between shrubland and grassland in southern Argentina, and (5) conversion during different periods between forest and secondary forest and conversion of grassland to agriculture in Veracruz. These changes indicate overall land-use intensification across all study areas during the interval studied. Loss of natural vegetation cover, namely forest and shrubland, is the most consistently observed change. Oaxaca presented by far the most complex patterns of change trajectories, with alternate exchanges between natural vegetation cover and bare ground or agriculture over the consecutive periods of time. Transformation to urban areas was less important as compared with other changes in all areas excepting in Chiapas. Similarly, expansion of tree plantations was relatively high in central Chile and southern Argentina, but was of lesser importance in other study areas than the changes indicated above.

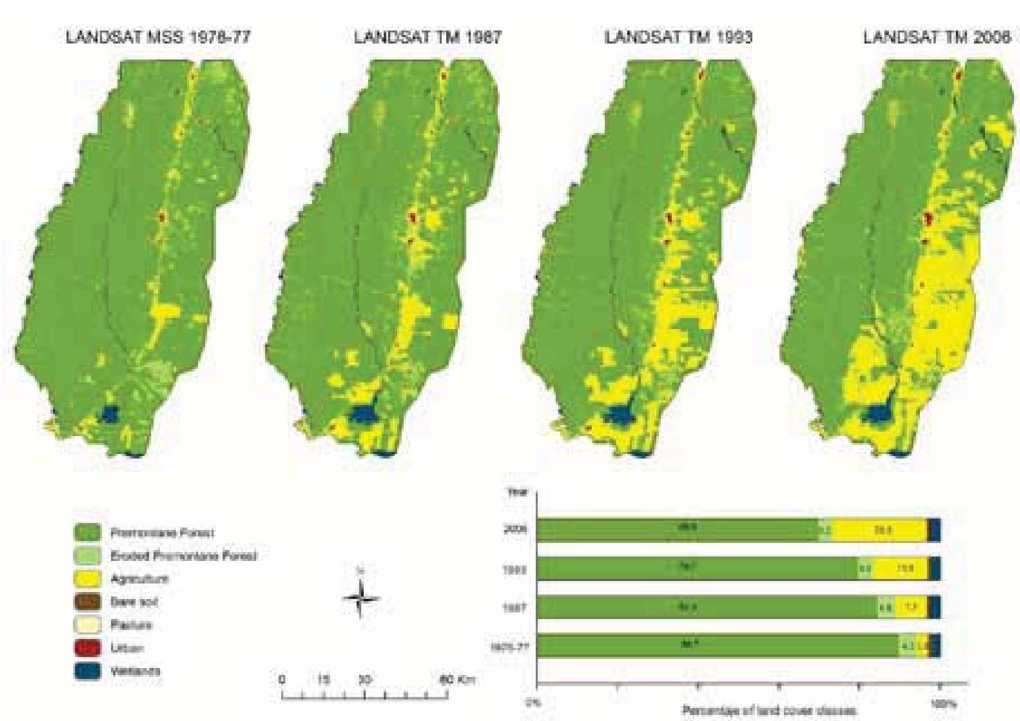


Figure 2.1 Land-cover maps based on classification of remote sensing imagery from the study area in northern Argentina for the years 1976–77, 1987, 1993 and 2006, and comparison of the respective extents of land-cover classes by percentage of study area (Cristóbal *et al.*, in preparation). Quantified maps of land-cover/land-use change and of current forest cover and forest loss for all study areas can be found in Rey Benayas *et al.* (2010a and 2010b).

Land use mosaic in La Sepultura Biosphere Reserve, Chiapas, Mexico. Photo: N. Tejedor

Vineyards in Casablanca valley, Chile. Photo: C. Echeverria

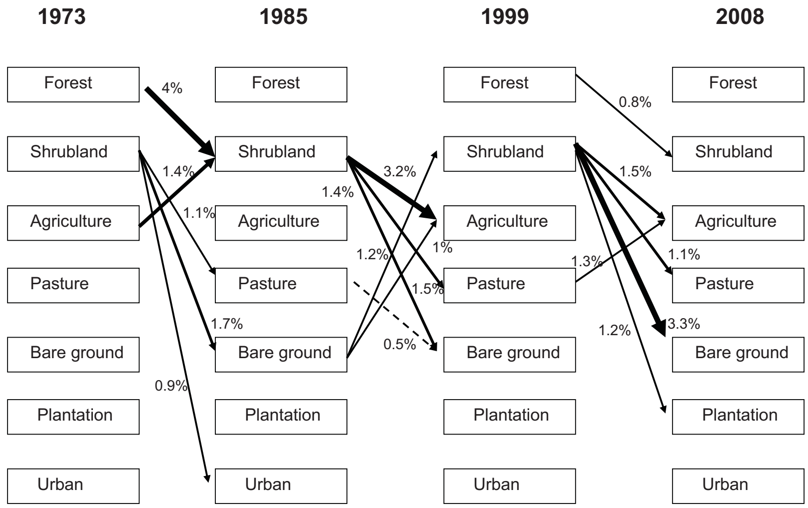
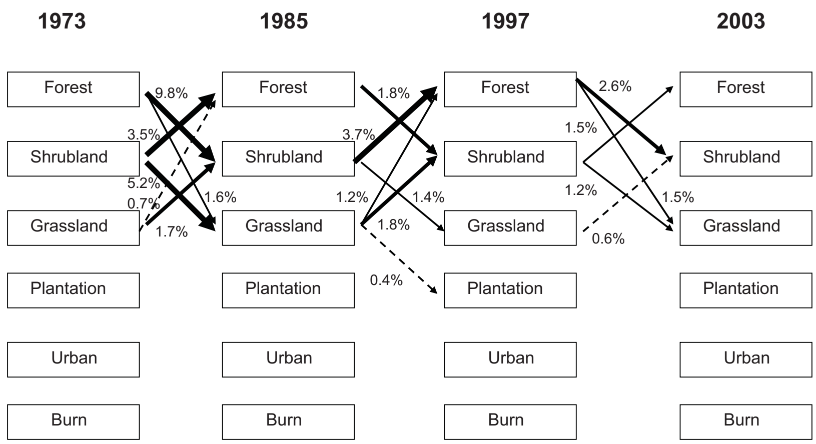


Figure 2.2a



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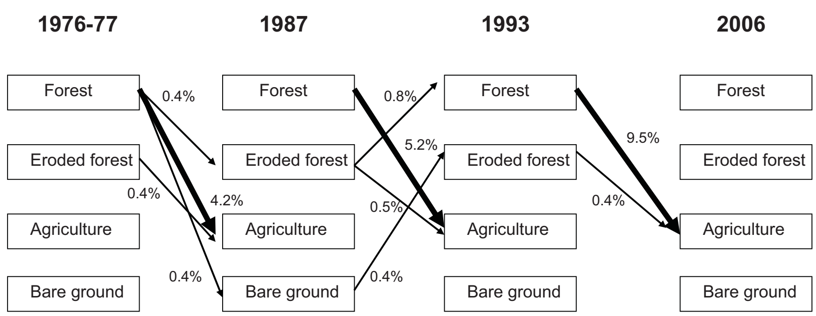


Figure 2.2c

Figure 2.2 Major change trajectories and their contributions to net change as a percentage of the study area in (a) central Chile (thick lines correspond to a net change of >3.2%, intermediate lines correspond to net changes between 1.6–3.2%, and thin lines correspond to net changes of <1.6%; only net contributions to change of >10,000 ha or 0.8% of the study area are represented) (Schulz *et al*., 2010); (b) southern Argentina (thick lines correspond to net change of >3.2%, intermediate lines correspond to net changes between 1.6–3.2%, thin lines correspond to net change of 0.8–1.6%, and dashed lines correspond to net change of 0.4–0.8%); (c) northern Argentina (thick lines correspond to net change >5%, intermediate lines correspond to net changes between 1–5%, and thin lines correspond to net change <1%; only net contributions to change >3200 hectares or 0.4% of the study area are represented); (d) Chiapas, Mexico (thick lines correspond to net change >0.3%, and dotted lines correspond to net change <0.1%), (0.1% of the study area corresponds to ca. 1600 ha); (e) Oaxaca, Mexico (thick lines correspond to net change >6 %, intermediate lines correspond to net changes between 6–1.6%, and thin lines correspond to net change <1.6; only net contributions to changes >350 ha. or 0.03% of the area are represented); (f) Veracruz, Mexico (thick lines correspond to net change ≥3.2%, and thin lines correspond to net change <3.2%; only net contributions to change > 2400 hectares or 1.6% of the study area are represented).

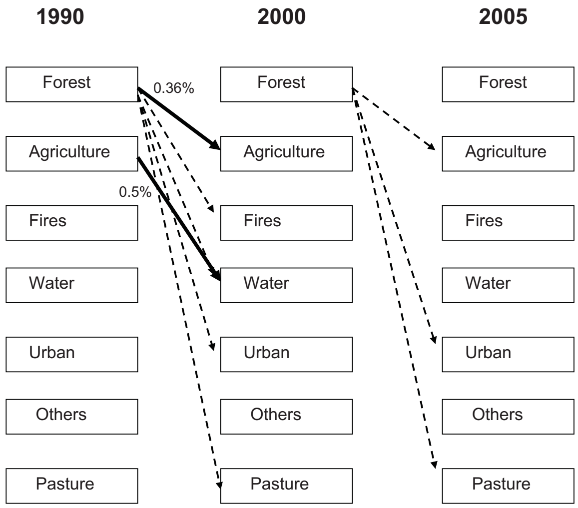
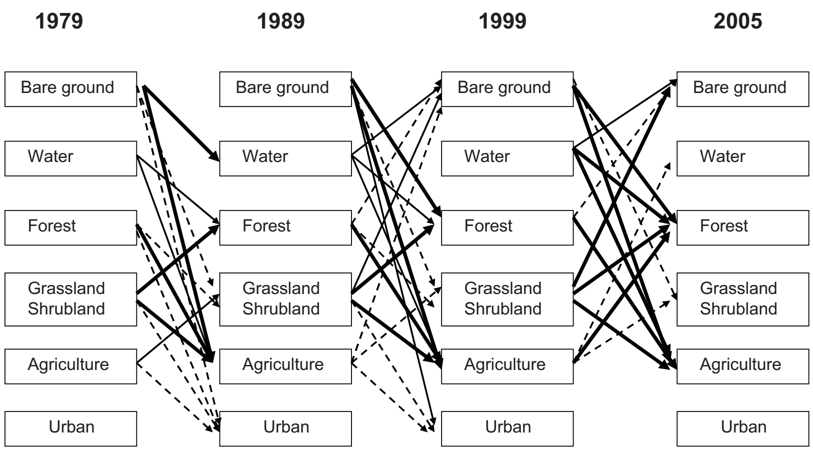


Figure 2.2d



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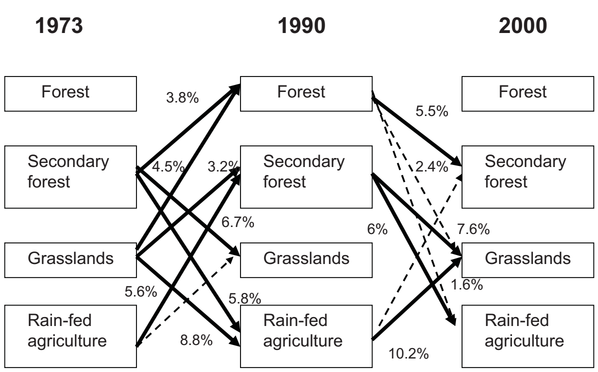


Figure 2.2f

Figure 2.2 (cont.)

Changes in land-cover/land-use types as inferred from mapping and quantification of remote sensing imagery and analysis of change trajectories among land-cover types indicate ongoing land-use intensification in all study areas. Major changes observed include a reduction in natural vegetation cover, namely forest and shrubland, and a strong increase in human-induced land-cover types such as cropland, pasture, bare ground, urban areas, and tree plantations with exotic species. Beyond this consistent pattern, the observed changes were of different intensity – when they existed at all – in the various study areas. The transformation of forest to the most highly influenced land-cover types, such as farmland, occurred with or without other intermediate land-cover types.

Land-cover change in the Mediterranean climate area of central Chile revealed a general trend towards a continuous reduction in forest and shrubland that, in turn, has led to an increase in provisioning ecosystem services such as food and timber production likely at the expense of biodiversity and hydrological services (Schulz *et al.*, 2010; **Box 2.2**). This process has involved a progressive modification from forest to shrubland vegetation, the predominant vegetation cover in this semi-arid landscape, and a relatively high loss of shrubland as a consequence of conversion to agriculture and timber plantations and, to a lesser extent, urbanization. This can be explained by an increase in local demand owing to population growth and an open market policy initiated after Chile's economic crisis at the beginning of the 1970s (Silva, 2004). The strong increase in agriculture has been stimulated by a combination of market liberalization, incentives for new export-oriented crops, introduction of new irrigation technologies, and improvements in road infrastructure (Valdés and Foster, 2005).

In contrast to central Chile, we observed an abrupt conversion of forest to agricultural land in northern Argentina, which has one of the highest agricultural land conversion rates in the country. In the 1970s, almost 95% of the area was covered by forests, and 90% by some type of tree cover. The deforested areas consisted mainly of pastures and rotational cropland plots in rural communities and small areas of premontane forests in areas where irrigation was possible. Since the 1980s, when soybean cultivation became highly profitable and began (Adelman, 1994; Brown and Malizia, 2004; Gasparri and Grau, 2009), huge areas of forest were converted at a rate of >20,000 ha/year (**Box 2.3**). Similarly in central Veracruz, Mexico, following a period of slight increase in forest area in the previous decade, forest area declined markedly in the 1990s with the establishment of powerful federal incentives, notably the Procampo programme, to promote the conversion of forest cover to cattle pasture and croplands (Klepeis and Vance, 2003; Montero-Solano, 2009). The programme originally applied to areas planted with beans, cotton, maize, rice, sorghum, soybeans, or wheat; however farmers supported by the programme could use their land for other crops, raising livestock or silviculture. Procampo is currently being phased out under the provisions of NAFTA (i.e. after 15 years) and forest cover appears to have begun to increase once again.

In Chile, there has been a pronounced expansion of timber plantations, mostly as a result of government subsidies for tree-planting that were introduced in 1974 and which stimulated the planting of *Pinus radiata* and *Eucalyptus globulus* (Aronson *et al.,* 1998). The expansion of timber plantations did not result in major conversions of native forest, as it did in southern Chile (Echeverria *et al.*, 2006) and the region in southern Argentina studied here. In this latter region, deforestation can primarily be explained by the occurrence of natural and anthropogenic fires which, in many cases, did not regenerate back into forests and remained as stable grasslands or shrublands (Mermoz *et al.*, 2005). However, dryland forest areas in this region are undergoing

Box 2.2 Vegetation cover change in mountain ranges of central Chile (1955–2008)

C. Villablanca, J. Hernandez, C. Smith-Ramírez, J. Schulz

The sclerophyllous forest is the most characteristic plant formation in the Mediterranean climate area of central Chile. Sclerophyllous trees and shrubs have hard leaves adapted to long dry summers and occasional morning frost. This plant formation is distributed mainly on south-facing slopes and creeks, where soil humidity is concentrated. The most common tree species at the dry end of the environmental gradient of Chilean sclerophyllous vegetation are quillay (*Quillaja saponaria*), maitén *(Maytenus boaria*) and litre (*Lithrea caustica*). In more humid places such as creeks and sheltered slopes, trees such as peumo (*Cryptocarya alba*), patagua (*Crinodendron patagua*), belloto del norte (*Beilschmiedia miersii*), pitra (*Myrceugenia exsucca*) and canelo (*Drimys winteri*) are found (Donoso, 1995).

Chilean sclerophyllous forests have high species richness and endemism (Villagrán, 1995). The high endemism and increasing degree of threat owing to extensive land-cover change have resulted in the inclusion of these forests as one of the 25 global biodiversity hotspots (Myers *et al.*, 2000). However, economic activities and increasing human population in central Chile have resulted in high impacts on natural resources, leading to important losses of biodiversity. Sclerophyllous forests have been eradicated and degraded over large areas of central Chile, especially in the central valley (see **Box 2.4**), but also in the Andean and coastal ranges. Deforestation and land use change are the result of expanding farmland and silvicultural plantations, as well as urban expansion (Armesto *et al.*, 2010; Schulz *et al.*, 2010). At the same time remnant stands are used for firewood and soil extraction, cattle grazing and trampling, and most importantly, subject to a high frequency of anthropogenic fires. Hervibory of shrub and tree seedlings by rabbits, horses and goats, is a major factor preventing sclerophyllous forest recovery on abandoned lands (Fuentes *et al.,* 1983).

The objective of this study was to assess the changes in land-cover that have occurred in the second half of the twentieth century in the Mediterranean-climate region of central Chile. We chose the period 1955–1975, as an antecedent to the patterns described by Schulz *et al.* (2010) for the period 1975–2008 in the same region. The study area was the landscape surrounding the Casablanca hills and valleys (2740.2 km^2, 32º50'00"–33º27'00" S and 71º36'00"–70º58'00" W (from sea level to 2190 m a.s.l.) and Cantillana hills (4304.18 km^2, 33°38'00"–34º15'00" S and 71°27'00"–70º38'00" W, from 145 to 2280 m a.s.l.), (**Fig. 1**). Plant species richness and composition were documented in sclerophyllous vegetation patches larger than 60 ha. We excluded extensive areas that had been deforested prior to 1955. Such areas are found south and east in the valley between Casablanca and the Cantillana hills. We estimated vegetation cover in 1955 using aerial photographs, and distinguished eight different cover classes: forests, shrubland, farmland, urban areas, bare soil, anthropogenic prairies, and exotic silvicultural plantations. Subsequently, we compared our results for 1955 with the vegetation cover map for 1975, obtained from Landsat images. We identified and quantified the major changes by cover type during this 20-year period.

To estimate land-cover changes between 1955 and 1975 we applied the Land Change Modeller in the IDRISI software programme; we used Landsat images to estimate land-cover changes between 1975 and 2008. The changes reported here are notable because shrublands in central Chile have been considered extremely persistent. The dynamics of land-cover change over these two decades of the twentieth century is shown in **Table 1**, **Figs. 2** and **3**. Between 1955 and 2008, 71,290.75 ha of forest were lost. From the total forest cover that existed in 1955, 64.2% (14,3160.96 ha) was below 700 m. Plant species richness is higher in Cantillana, and in Casablanca below 600 m (Universidad de Chile, 2009). However, the areas of greatest species richness were replaced by other land-cover types by the first half of the twentieth century or earlier.

The percentage of change in sclerophyllous forest cover from 1955 to 1975 was comparatively low compared to the period 1975 to 2008, where natural vegetation cover was lost much more rapidly (Schulz *et al.*, 2010). In the study areas, between 1955 and 1975, forest cover decreased by 8.5%, however between 1975 and 2008, forest cover decreased by 45%. Between 1955

Box 2.2 (cont.)

and 1975 Matorral (shrubland cover) increased by 5.5%, however, it decreased by 22.7% in the period between 1975 and 2008. This was probably the result of the expansion in agricultural land and the development of an agricultural export industry in Chile coinciding with the opening up of international markets. The development of new methods of irrigation during the 1980s was responsible for the expansion of cultivated land up to altitudes of 500 m on coastal hills; this affected the plant diversity of many remnant patches of sclerophyllous forest and shrubland.

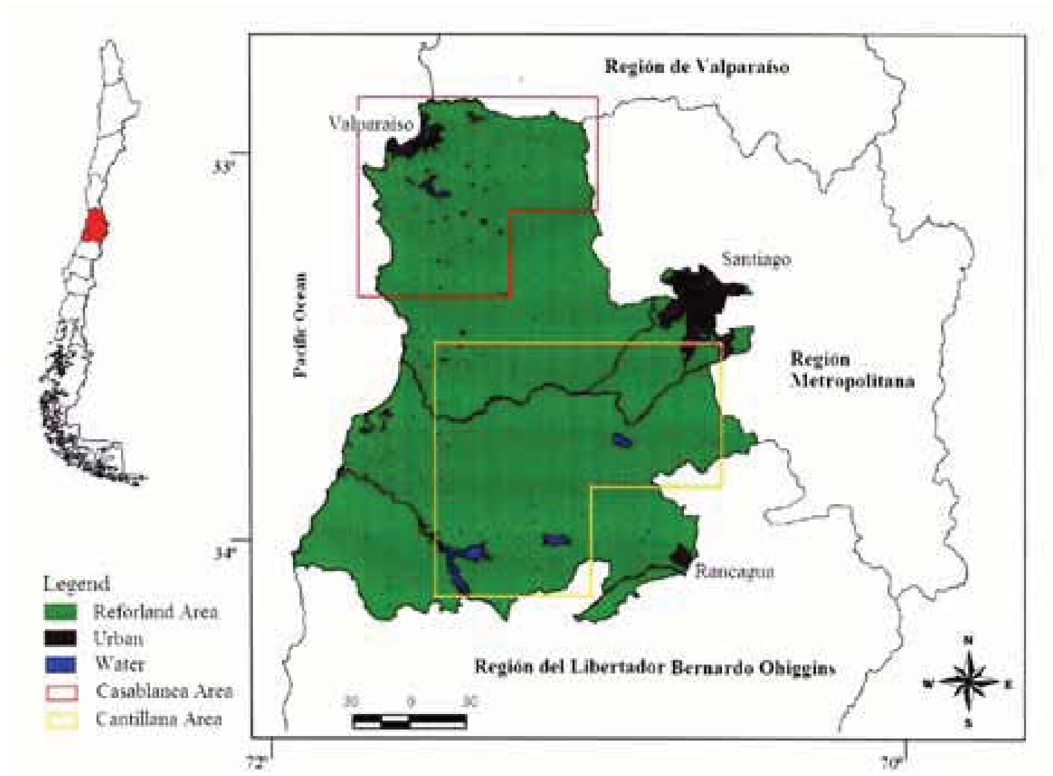


Figure 1 Area of study in the Casablanca and Cantillana hills (red and yellow lines indicate the area mapped).

Table 1 Dynamics of land-cover change in hectares on the Casablanca (CB) and Cantillana (C) sites.

Ha	Forest	Shrubland	Agricul-tural land	Urban soil	Bare soil	Water	Prairie	Plantations
CB 1955	59,905.51	148,625.93	9,634.07	5574.85	3120.80	1690.97	39,440.08	5031.51
CB 2008	25,549.62	104,176.17	28,550.01	17,471.1	34,329.55	801.46	39,043.49	20,238.92
C 1955	83,255.45	147,079.46	119,254.14	3062.85	14,056.02	1425.32	44,688.32	401.11
C 2008	46,320.59	136,973.54	133,873.81	6971.57	42,607.54	4115.66	32,685.24	578.71

Box 2.2 (cont.)

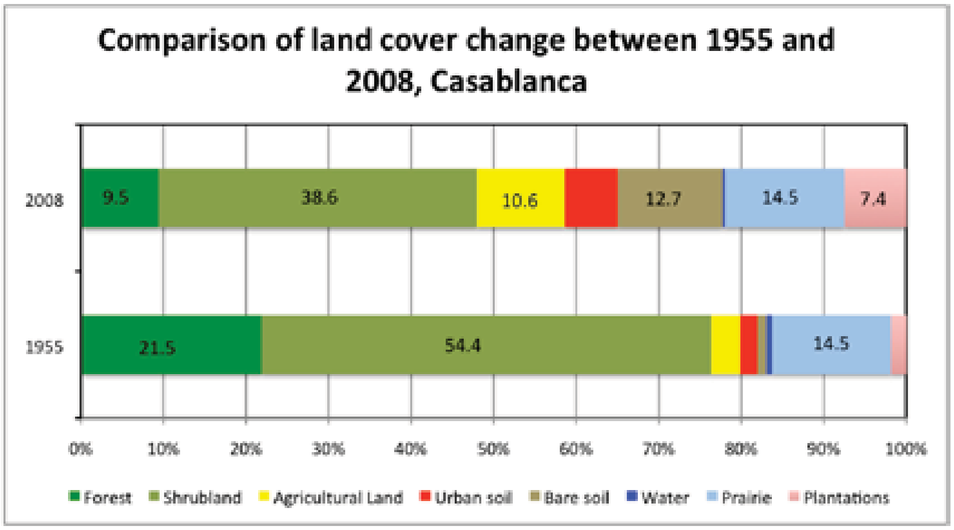


Figure 2 Land-cover change in Casablanca between 1955 and 2008.

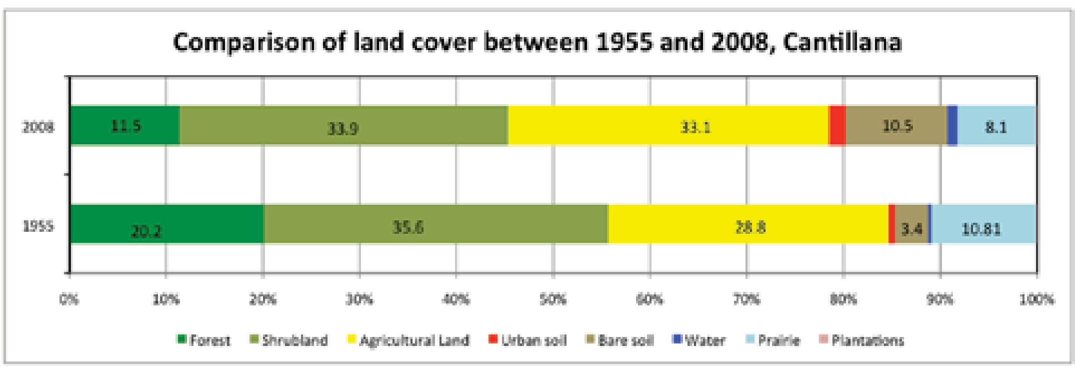


Figure 3 Land-cover change in Cantillana between 1955 and 2008.

Box 2.3 Land-use change in the Yungas Biosphere Reserve and its area of influence, Argentina (1975–2008)

S. Pacheco, L. Cristóbal, K. Buzza

The Biosphere Reserve of the Yungas (RBYungas) was created in northwestern Argentina in 2002 under the UNESCO Man and the Biosphere (MAB) programme. It is one of the largest reserves in the country, extending over approximately 1,350,000 ha, and includes two provincial territories (Jujuy and Salta). The reserve was created as part of efforts to implement actions conducive to conserving and managing the Yungas region (subtropical mountain forests) in a sustainable way.

The RBYungas includes mainly subtropical mountain forests, particularly in the northern latitudinal sector, which are functionally connected to the central sector of the Yungas and to the Chaco forests in the surrounding areas. As a result of agricultural activity, these environments are being fragmented, with remaining fragments sometimes connected by areas that act as corridors. The objective of this study was to describe the land-use change process in the northern and central sectors of the Yungas and its transition to Chaco, with special emphasis on the RBYungas for the period 1975–2008.

Box 2.3 (cont.)

The identification of transformed areas was carried out through visual interpretation of Landsat satellite images. We developed a time series of an area of more than 5 million ha located in Jujuy and Salta. The years included were 1975, 1985, 2005 and 2008. For the four years analyzed, we calculated the total deforested area and the annual transformation rate. For each year of the time series, we identified types of crops, determined their slope and established the ecoregion that was transformed in each case. For the RBYungas we determined the deforested area for each year analyzed and the remaining surface which may be subject to transformation.

From 1975 to 2008, the transformed surface in the study area increased almost 13% (**Table 1**). During the 1970s, the transformed areas were mainly concentrated in the flat areas of the premontane forest, on the west side of the study area. During the 1980s, the expansion of the agricultural frontier began in the east of the region, mainly in the province of Salta, occupying the Chaco environments (**Fig. 1**).

Figure 1 Spatial distribution of transformed land in the study area and RBYungas for the years 1975, 1985, 2005 and 2008.

The productive activities carried out in the 1970s include sugar cane, tobacco and agricultural cultivation. During 1985 these three activities remained important but the production of soybean combined with beans represented almost 30% of production. During 2005, these two new categories accounted for almost 50% of production. During the four years analyzed, 90% of the transformation occurred on slopes below a 5% gradient. The transformation of the forest in steeper areas was mainly for forest plantations, pastures and agricultural plots of small size.

Box 2.3 (cont.)

Table 1 Transformed area in different years and annual transformation rate for the study area during the period 1975–2008.

Year	Transformed (ha)	% of RBYungas	Annual transformation rate (ha)
1975	359,143	6,7	
1985	611,295	11,5	25,215 (1975-1986)
2005	908,845	17	14,878 (1998-2004)
2008	1,034,486	19,4	41,880 (2004-2008)

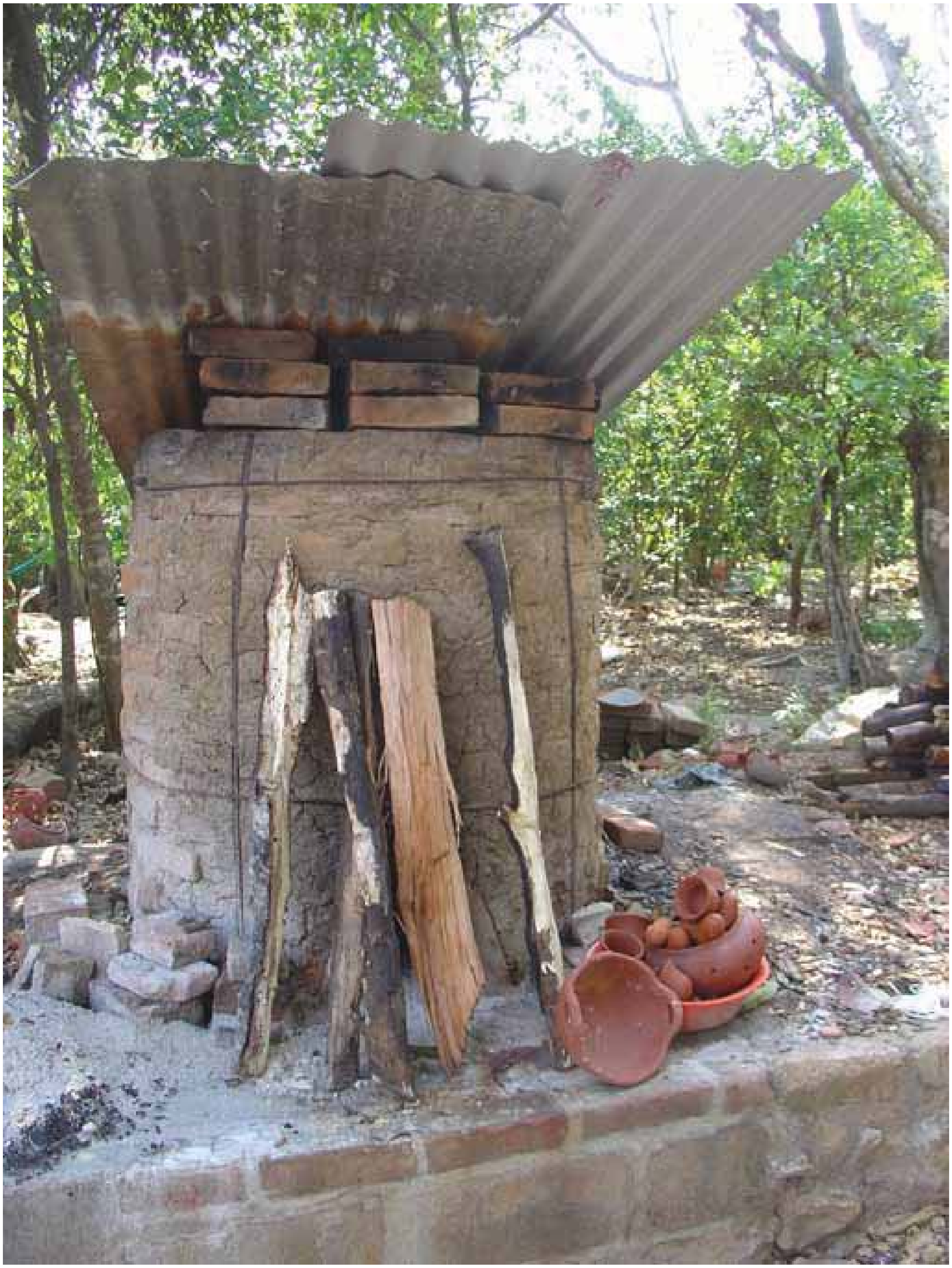
Transformation within RBYungas

In 1975, 4% of the RBYungas was transformed into agricultural land, mainly concentrated in flat areas; this reached 6.5% in 2008. This represents an annual transformation rate which increased from 930 ha for the period 1975–1985 up to 3274 ha between 2005 and 2008 (Table 2). Almost 90% of the RBYungas surface is represented by steep areas with a slope of a gradient of more than 5%. Only 150,000 ha correspond to flat areas suitable for agricultural use. If we analyze the transformed area taking into account only available flat areas within the RBYungas, we observe that during 2008 more than 50% of this surface was transformed (Table 2).

Table 2 Transformed surface in different years and annual transformation rate for the RBYungas during the period 1975–2008.

Year	Transformed (ha)	% of RBYungas	% flat areas in RBYungas	Annual transforamtion rate (ha)
1975	53,597	4.0	34.2	
1985	62,725	4.7	39.4	930 (1975-1986)
2005	77,164	5.7	49.2	722 (1998-2004)
2008	86,987	6.5	55.3	3.274 (2004-2008)

These results indicate that the transformation of premontane forest has been increasing in recent years. This process began in areas with slopes of less than 5% gradient with the planting of sugar cane which requires irrigation; it resulted in the near disappearance of forests in flat areas at a regional level. Since the 1980s, there has been an expansion of the agricultural frontier towards drier areas of the Chaco owing to changes in technology, an increase in precipitation in the last century and the incorporation of the soybean crop which does not require irrigation. More than 50% of the flat surface of the RBYungas has already been transformed into agricultural land. RBYungas remains connected to its surrounding natural areas through those slopes that remain forested. Currently, local governments have implemented land-use plans, which delimit production areas and areas that must be protected in order to maintain the functional connection between the different environments of the region.



Firewood next to a potter's kiln, Chiapas, Mexico. Photo: B. Ferguson

a small but steady conversion to plantations with exotic conifers. In southern Argentina, rapid expansion of urban areas coincided with the abolition of the urban limits by the Ministry of Housing and Urbanism in 1979 and the liberalization of the urban land market (Kusnetzoff, 1987).

Changes in forest extent

Forest loss was consistently detected in all study areas, ranging from an annual rate of –1.7% in central Chile to a negligible –0.12% in the Central Valley of Chiapas, with an average rate of –0.78% across all study areas (**Table 2.1**). Some study areas have experienced a relatively high proportion of forest loss (15% to 9% of the study area in central Chile, 90% to 70% in northern Argentina, and 11.3% to 6.56% in central Veracruz), whereas others have experienced relatively little deforestation (13% to 11% in southern Argentina, 59% to 57% in Oaxaca, Mexico) or hardly any changes in their forest extent (32% of the study area in Chiapas at both reference dates, as 68% of forest land had already been lost in this region by the beginning of the study period).

Table 2.1 Percentage forest cover detected in the 1970s (1990 for Chiapas*) and mid-2000s (see Table 1 Box 2.1 for exact dates) and annual deforestation rates in each study area. **Refers to undisturbed forest cover in the 1973–2000 period when images are all Landsat and thus have more comparable spatial resolutions.

Study area	% Forest cover in early or mid-1970s	% Forest cover in 2000s	% Annual rate of change
Central Chile	43.3	33.9	–1.7
Southern Argentina	17.3	16.4	–0.17
Northern Argentina	94.0	73.0	–1.3
Veracruz (Mx)	11.3	6.56	–1.22**
Oaxaca (Mx)	59.3	56.6	–0.18
Chiapas (Mx)	32.1*	31.5	–0.12

However, forest loss varied considerably between the analyzed time periods in most study areas. Thus, northern Argentina showed high forest loss (>5%) between all analyzed periods, central Chile lost forest extent in all periods except from 1985 to 1999 when there was hardly any change, and central Veracruz gained total forest cover at a yearly rate of 0.6% between 1973 and 1990 but then underwent considerable forest loss at a yearly rate of 4.33% between 1990 and 2000.

The analysis of change trajectories revealed that the patterns of conversion of forest cover also differed between study areas. The conversion of forest to agriculture was mediated by an intermediate shrubland state in central Chile (**Fig. 2.2**) and southern Argentina, whereas in Veracruz, a similar trend was observed for the conversion from primary forest to agriculture mediated by an intermediate secondary forest state. However, in northern Argentina and Oaxaca, and to a lesser extent in Chiapas, most forest loss occurred as a result of direct transformation into agricultural land. In all study areas, loss of forest extent was partially mitigated by forest re-growth as a result of farmland abandonment (**Fig. 2.3**).

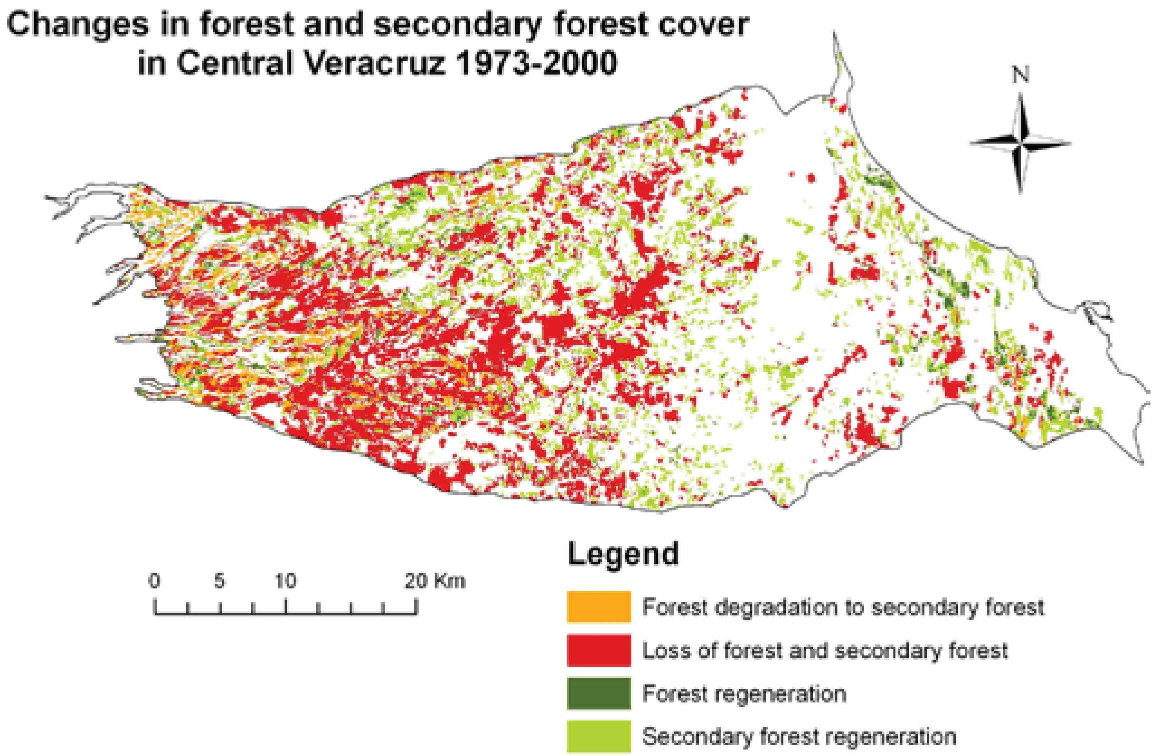


Figure 2.3 The extent and distribution of primary forest at a given time is a balance between forest loss and degradation and forest regeneration. This figure illustrates the distribution of these different processes for the case study landscape in Veracruz, Mexico, for the period 1973–2000. Similar maps for all study areas can be found in Rey Benayas *et al.* (2010b).

While forest loss was a consistently observed outcome of land-use intensification, annual rates of deforestation were highly variable between study areas and among different time periods within the same study area. The highest deforestation rate was observed in Chile; nevertheless, this rate is relatively low compared to rates in temperate forests in south-central Chile (Echeverria *et al.,* 2006). This is probably attributable to the fact that central Chile has been densely populated since early European colonization and major conversions of forest cover had taken place long before the 1970s (Conacher and Sala, 1998; **Box 2.4**). Historical data from central Veracruz suggest that dryland forest cover has been suppressed (<33% cover) at least since the start of the previous century when land reforms were implemented following the Mexican revolution (**Box 2.5**). Similarly, most deforestation in Chiapas occurred long ago (Challenger, 1998) and has been relatively minor in recent times (**Box 2.6**). Our results suggest that this region has undergone a comparable level of historical deforestation to other dry forest ecoregions in Mexico. Challenger and Dirzo (2009) reported that dry forest loss at the country level between 1976 and 1993 amounted to 177,000 ha per annum (annual deforestation rate of 1.6%) and reduced to 44,416 ha per annum (annual deforestation rate of 0.5%) over the next decade (1993–2002). Deforestation rates found in our study area in Oaxaca were slightly lower as compared with other areas with similar vegetation in Oaxaca and in Mexico (Aguilar *et al.*, 2000; Velázquez *et al.*, 2003; Díaz-Gallegos *et al.*, 2008). Cayuela *et al.* (2006), however, reported an annual deforestation rate of 4.8% for the highlands of Chiapas in the period 1990–2000.

Primary forests are lost because they are directly converted to cropland or grazing land or because they are degraded by permanent grazing pressure, firewood collection and charcoal

Box 2.4 Historical distribution of the dryland forest in central Chile during the Spanish conquest in the 16th century

C. Echeverria, R. Fuentes, R. Torres, P. Camus.

Historic landscape reconstruction through documentary sources is useful to (i) assess the dynamics of historical land-use changes, (ii) define the potential of dryland forest ecosystems, and (iii) place resource management practices of indigenous people and other local communities in a historical context (Prieto *et al*., 2003). The coastal zone in central Chile that extends from the cities of Santiago to Valparaiso presents evident signs of environmental degradation that has occurred since the 16th century. Traditionally, Chilean historians have presented an idyllic image of a territory covered by beautiful forests (Barros Arana, 1884), while others, more recently, have provided information indicating an absence of forests over large areas in historic times (Camus, 2002). Primary documentary sources offer a valuable description of the past as they narrate the first impressions of European colonists when they laid eyes on the landscape for the first time.

The reconstruction and mapping of dryland forest in the 16th century, at the onset of European colonization, is key to a clear understanding of the current and historical patterns of land-use change and the development of restoration strategies in central Chile. The objective of this exercise was to reconstruct a picture of the vegetation between Santiago and Valparaíso through a spatially-explicit approach that integrates information from documentary sources and environmental factors into a GIS.

Visual descriptions of vegetation were obtained from field notes taken from travellers through the region. These descriptions were collected from reviews of primary and secondary historical sources written in the 16th century. Most of the visual descriptions used in the present study were gathered in the Casablanca valley, Colina zone and the Aconcagua River route. The main primary documentary sources used in this study were drawn from registries maintained by Santiago's Cabildo (town council), land measurements and registries, chronicles, letters and travel diaries. However, further descriptions from secondary documentary sources were also used when they clearly referred to descriptions of vegetation that existed before the arrival of the Spanish. Visual descriptions of naturalists such as Charles Darwin and Edward Poeppig were used to model species' distributions on the Aconcagua-Valparaiso route and its surroundings. We discarded those descriptions that did not have a precise enough spatial reference. As a result, we did not use Claudio Gay's botanical descriptions.

The descriptions and roads used by travellers were spatially plotted on a 30 m-resolution elevation map. Then, environmental requirements such as aspect and elevation of the description points in relation to vegetation composition were obtained through a review of the literature (Donoso, 1982; Donoso, 1995). This information was used to generate digital maps of habitat suitability for each category of vegetation (group of species or individual species in some cases) through a combination of ranges of elevation with categories of aspect (Table 1). North and south aspects are the major environmental factors that determine patterns of species distribution in dryland landscapes in this part of Chile (Donoso, 1995). South-facing hills are characterized by lower levels of solar radiation/insolation, and therefore higher humidity in the soil and air than north-facing sites.

In some cases the documentary sources provided specific locations for some currently threatened species such as *Jubaea chilensis* (Grau, 2004) and *Porlieria chilensis*. This enabled the historical distribution at the species level to be mapped. Similarly, several historical descriptions mentioned the abundant presence of *espinales*, a disturbed pseudo-savannah dominated by *Acacia caven*, across the study area. This enabled different sub-categories of *espinales* to be mapped. Additionally, documentary sources containing detailed descriptions of some of the current main cities in the study area such as Santiago, Valparaiso and Quillota were mapped.

Box 2.4 (cont.)

Table 1 Equivalence of meanings and historical descriptions for each detected vegetation category.

Vegetation type	Altitude (m a.s.l) and aspect	Sample of original texts described by travelers during the 16th century	Historical references
Grassland	> 200 North and East	"Las únicas plantas que cabría mencionar eran arbustos pequeños e inaperentes, que ostentaban sólo de vez en cuando todavía en temporada desfavorable una miserable flor y que se presentaban semisecos y polvorientos"	Poeppig, E. 1835
Open espinal	200 – 350 North and East > 300 South and West	" … Todos los campos estaban desiertos, solo se veían cubiertos de ciertos arboles espinosos que hacen mui incómodo el camino".	Frezier, M. 1716
	350 – 450 North and East 300 – 350 South and West	" … me encuentro otra vez en tierras salvajes cubiertas en forma muy rala por acacias y algarrobos cuya compañía, empezaba yo a sospechar, no iba a perder en esta tierra"	Schmidtmeyer, 1824
Dense espinal	350 – 450 North and East 300 – 350 South and West	" … Numerosos troncos mejor talados a uno o dos pies del suelo, donde ensanchaban su espacio estéril no parecían mejorar su perspectiva, pero indicaban que este valle había estado cubierto tupidamente por ellos antiguamente … "	Schmidtmeyer, 1824
	350 – 450 North and East 300 – 350 South and West	" … noto sobre la vertiente septentrional no crecen sino zarzas … "	Darwin, C. 1839
Sclerophyllous forest	350 – 450 North and East 350 – 1000 South and West	" … He visto algunos lugares bonitos, que consisten en pequeñas colinas y cañadas de formas suaves, cubiertas de varias clases, mas vegetación que la vista hasta ahora y de verdor ás agradable"	Schmidtmeyer, 1824
	350 – 450 North and East 350 – 1000 South and West	Santiago " … es un hermoso y grande llano como tengo dicho.Tiene a cinco y seis leguas montes de muy buena madera que son unos árboles muy grandes que sacan muy buenas vigas. hay otros árboles que se llaman canela"	Gerónimo de Bibar, 1558
	350 – 1000 South and West	Estero de Pocochay como "Maquilemu", o "bosque de maquis".	Ginés de Lillo, 1605
	350 – 1000 South and West	" … la vertiente meridional está cubierta de un bambú que llega a alcanzar hasta15 pies de altura".	Darwin, C. 1839

Box 2.4 (cont.)

Table 1 (cont.)

Vegetation type	Altitude (m a.s.l) and aspect	Sample of original texts described by travelers during the 16th century	Historical references
Jubaea chilensis	500 – 2000 North	"Hay palmas y solamente las hay en esta gobernación en dos partes, que es en el río de Maule hay un pedazo que hay de estas palmas, y en Quillota las hay en torno de siete y ocho leguas"	Gerónimo de Bibar, 1558
	500 – 2000 North	"En algunos lugares se encuentran palmeras y quedo muy asombrado al hallar una de ellas a 4.500 pies de altitud … "	Darwin, C. 1839
Indigenous settlements	>200	"La población vivía sobre todo en ese Sector, llamado también 'Camino de Coquimbo', pues este seguía por Calera a la cuesta del Melón y se juntaba en La Ligua con el de la costa, para continuar al Norte".	Keller, Carlos, 1960

The reconstruction (mapping) of the vegetation revealed that there was once a greater number of species spread over a greater area (**Fig. 1**). In the 16th century, sclerophyllous forest occupied an area of approximately 115,000 ha, mainly south facing. *Espinales* covered approximately 670,000 ha on north-facing and flat sites. At the municipal level, 21% of the total area in Casablanca municipality was occupied by sclerohyllous forest and 66% by *espinales*. In Quilpué, these values were 18% and 68%, respectively; and in Melipilla, 34% and 67%, respectively.

At the species level, the reconstruction of *J. chilensis* distribution revealed that this once covered approximately 17,000 ha, a much larger area than at present (**Fig. 1**). The species was distributed in two main populations, one on the coast around the current city of Viña del Mar, and the other in the La Campana mountains (**Fig. 1**). *Nothofagus macrocarpa* forest covered most of the summit on the Roble hills and the Cantillana hills (**Fig. 1**), with a total area of 33,800 ha. Towards the east, *Porlieria chilensis* occupied an area of approximately 5,100 ha of the hills around the Santiago area (**Fig. 1**).

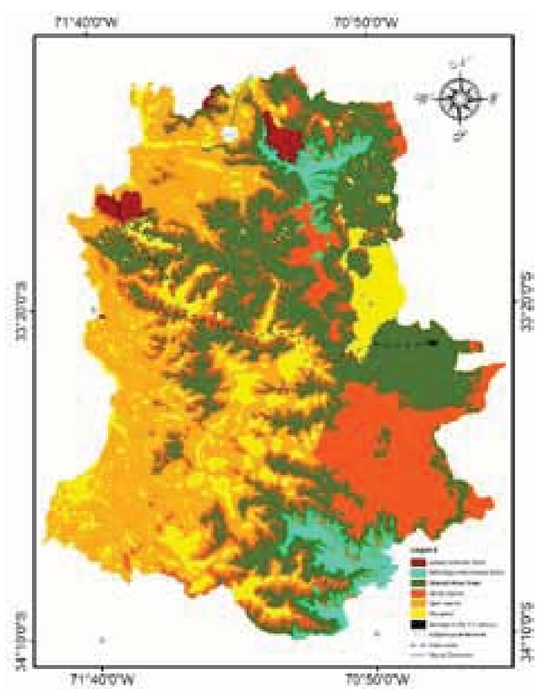


Figure 1 Distribution of vegetation types and human settlements during the Conquest in the 16th century.

Box 2.4 (cont.)

Although the area of forest coverage in 16th century was larger as compared to the present day, in the eyes of the newly arrived Spanish population of central Chile, tree species were already scarce at that time. The Cabildo therefore issued several directives regulating the cutting of trees. One of the main indigenous settlements in the 16th century was established in the current Quillota valley and occupied approximately 1,800 ha (**Fig. 1**). Santiago was the main Spanish settlement during the Conquest, extending over 320 ha (**Fig. 1**). Some documentary sources also provide evidence of the main disturbances affecting vegetation in central Chile in the 18th century after the founding of Santiago. The establishment of Santiago resulted in the expansion of urban areas and rangelands for livestock into the Central Valley during the 18th century. This expansion resulted in the high consumption of tree and shrub species (Cunill, 1995).

Our results demonstrate that by the 16th century, the landscape in central Chile was dominated by different vegetation types. Sclerophyllous forest and some species were more abundant on south-facing slopes, while *espinales* covered large areas on the north-facing and flat sites across the study area. The presence of this human-induced vegetation type reveals that the original vegetation had already been disturbed by indigenous people. All this indicates that dryland vegetation has been profoundly and irreversibly transformed since the Conquest until the present time, highlighting the need for ecological restoration. The historical analyses presented here can be used to inform restoration plans.

Box 2.5 Historical reconstruction of land-use patterns from 1920 to 1960 on communal lands of Paso de Ovejas, Veracruz, Mexico

J. Ortiz, F. López-Barrera, J. Callejas, R.H. Manson

The primary forests in the municipality of Paso de Ovejas suffered few alterations following the Spanish conquest. However, in the 19th century these lands came under the management of the military, merchants and foreign investors. Most forest cover in the lower sections of the municipality, where irrigation was increasingly prevalent, were subsequently replaced by commercial plantations such as sugarcane. Conversely, the highlands, hills, slopes and canyons, which were largely isolated from human infrastructure and roads, suffered relatively few alterations to their natural vegetation cover until the first decades of the 20th century, when they were used as pastures.

Increases in human populations were also a contributing factor to these changes. In 1799, census data registered a population of barely 100 people. A century later this number had increased to 3572. The heads of family included men who worked on the land of the large private properties (*haciendas*) as labourers, sharecroppers, and day labourers. Each family had permission to use a small part of the cultivated land to grow staples such as corn, beans and chillies, and to raise domestic animals such as pigs or chickens.

After the Mexican revolution and the establishment of the Constitution of 1917, these *haciendas* were largely dismantled as part of agrarian reforms. Initially, the most productive lands were transformed into communal lands known as *ejidos* and divided among the peasants that had worked in the *haciendas* for many years. Under the *ejidos* systems land was kept in collective trusts for the peasant communities who were allowed to use it for farming and natural resource extraction. In the first thirty years of the 20th century, the population of Paso de Ovejas doubled to more than 7350 people. This population growth continued during the next two decades. By the 1970s, owing to both intrinsic population growth on communal lands and the arrival of new immigrants, the population had doubled again reaching a total of 15,271 people. The greatest increases were observed among the male population, specifically the 25 to 29-year age group.

Box 2.5 (cont.)

The combination of rapid population growth on communal lands, including those used for cultivation and cattle ranching, and those considered too infertile for agriculture, put considerable strain on this new form of land tenure. The *ejido* system is one of the main legacies of the Mexican revolution; it was incorporated in the 1917 Constitution. *Ejido* members lived in communities. They were designated land for housing with separate parcels of land designated for cultivation. As family sizes increased, the parcels of land they were assigned came under increasing pressure resulting in intensification and transformation of land uses. Forests came under particular pressure as they were felled for fuel or lumber, and in order to make room for crops, and cattle pastures.

In addition to these changes in land ownership and population demographics, the conversion of forest cover to other land uses was actively promoted by public policies, and related financial and technical assistance provided to *ejidos* by the State and Federal governments. In 1920 the Law for Idle Lands was established nationally and triggered large-scale deforestation across Mexico. The objective behind this law was to increase the volume of crops to feed the rapidly growing Mexican population. This was followed by the creation of the National Ejidal Credit Bank (1935), the organization of farmers in the National Peasant Confederation (1938), the creation of the Agriculture Bureau and Mexican Fertilizers Bureau (1943), the establishment of the National Agriculture Plan (1953), and the formation of the National Seed Producers Organization (1960), all of which facilitated the conversion of forest cover to other land uses.

As a result, remaining forest cover was increasingly limited to inaccessible lands (steep slopes or rocky soils far from roads and towns). Land-use patterns for 23 *ejidos* in the municipality of Paso de Ovejas from 1927 to 1968 are described in Table 1. These data were obtained through the revision of historical documents and maps from the National Agrarian Registry in Mexico (Registro Agrario Nacional). Forest cover was found to be absent (13 *ejidos*) or limited (from 19% to 35%) in most *ejidos* with only two showing forest coverage of more than 40% of their land area (mainly secondary forests; Table 1).

This study showed that in some tropical areas in Mexico, most deforestation, degradation, and fragmentation of forest cover probably occurred prior to 1920. These patterns of land-use change are largely undetectable using current satellite-based methods and imagery data available since the early 1970s. For example, in Paso de Ovejas, the extension of primary forest was only 2.06% in 1973 and increased to 6.28% in 1990 (Montero *et al*., in prep). These results highlight the importance of a long-term historical perspective for understanding and interpreting current patterns of land use and the overall impact of public policies on patterns of land-use change in the region.

Table 1 Historical records of type of land use present on 23 communal lands (*ejidos*) in central Veracruz, Mexico, including the year when they were created, area, and percentage cover of different types of land use.

Ejido name	Year	Extension (ha)	Rain-fed agriculture	Irrigated agriculture	Grasslands	Primary and secondary forest	Urban
Acazónica	1927	1610	14.7		72.9	12.4	
Tierra Colorada	1928	150	100				
Plan de manantial	1928	312	79.2		20.8		
Palmaritos	1928	482	24.1	52.5	23.4		

Box 2.5 (cont.)

Table 1 (cont.)

Ejido name	Year	Extension (ha)	Rain-fed agriculture	Irrigated agriculture	Grasslands	Primary and secondary forest	Urban
Paso de Ovejas	1930	1617	100				
Bandera de Juárez	1930	472	11.0		10.6	78.4	
Loma del Nanche	1930	262	64.9			35.1	
Puente Jula	1930	168	100				
Cerro de Guzmán	1931	359	54.3		21.4	24.2	
Mata Grande	1931	100	76.0		24.0		
Paso Panal	1932	370	100				
Patancán	1932	180	100				
La Víbora	1935	279	100				
Cantarranas	1935	1124	66.2			33.8	
El Angostillo	1936	900	80.0			20.0	
Cocuyo	1936	428	80.4			19.6	
El Mango	1937	211	100				
Mata Mateo	1937	714	52.9			47.1	
Mata Grande	1941	260	67.7		32.3		
Rancho Nuevo	1958	630	82.5		15.9		1.6
El Angostillo	1964	900	75.2			22.6	2.2
Acazónica	1964	2182	48.1		40.6	9.2	2.1
Loma del Nanche	1968	156			100		
Average		**602.87**	**68.57**	**2.28**	**15.74**	**13.15**	**0.26**
Standard deviation		**554.98**	**31.17**	**10.94**	**25.85**	**20.09**	**0.69**

***Austrocedrus chilensis* stand, Nahuel Huapi, Argentina. Photo: A.C. Newton**

Shrubland and steppe vegetation, Nahuel Huapi, Argentina. Photo: J. Birch

Box 2.6 Tuning up coarse-grained potential vegetation maps for estimation of historical forest loss in tropical Mexico

R. Vaca, L. Cayuela, J.D. Golicher

In areas with a long history of disturbance, historical forest loss is a major issue. Most deforestation in these areas has occurred prior to the development of remotely sensed techniques. In such circumstances potential vegetation maps can be used as a baseline for the estimation of historical forest loss (e.g. Trejo and Dirzo, 2000), as they represent the area hypothetically covered by forest in the absence of human disturbance (Bredenkamp *et al*., 1998; Moravec, 1998). However there is a general concern that the resolution of most of the available maps of potential vegetation is too coarse for real-world applications (Bredenkamp, et al., 1998; Hartley *et al.*, 2004).

A typical method for constructing a potential vegetation map involves identifying remnants of vegetation with natural or near-natural character (Zerbe, 1998). The vegetation found in these remnants may be assumed to potentially extend to a wider geographical area with similar environmental conditions (Moravec, 1998; Zerbe, 1998). Map inaccuracies often result from the coarse resolution in the available maps of predictor variables (van Etten, 1998). Coarse scale maps may overlook variability in mountainous and other areas in which fine scaled climatic gradients determine the observed vegetation type (Franklin, 1995). In order to use vegetation maps effectively their resolution must be adjusted to the needs of pure and applied biological and ecological research (Araújo *et al.*, 2005; McPherson *et al.*, 2006). For many applications, data at a fine (≤1 km^2) spatial resolution are necessary to capture environmental variability that can be partly lost at coarser-grained resolutions (Hijmans *et al.*, 2005). A particular challenge is thus the generation of a fine-grained map of the potential vegetation over a large area. Climate is widely known to condition the formation of different vegetation types (Woodward, 1987). Thus, when the grain of potential vegetation maps is coarser than that of climatic layers, one possible solution is to use climate to downscale potential vegetation maps through statistical modelling.

Here, we illustrate the usage of climatic information in the downscaling of coarse-grained potential vegetation maps with reference to the state of Chiapas, a region historically affected by human activity located in Southern Mexico. The aim of this study was to define the original distribution of the area occupied by different tropical vegetation types in the region. At present, one widely recognized source of information on the potential distribution of vegetation types of Southern Mexico is Rzedowski's potential vegetation map (1990) (Olson *et al.*, 2001). This is represented at a scale 1:4,000,000, which is clearly limited as a baseline for the estimation of historical forest loss (Trejo and Dirzo, 2000). In response to this problem, we downscaled Rzedowski's potential vegetation map for Chiapas, from a 1:4,000,000 to 1 km^2 grid resolution, using climatically-based random forests models.

To obtain categorical values for the dependent variable (classes of potential vegetation), we systematically extracted points at one km distance from Rzedowski's potential vegetation map. We did not obtain samples from aquatic vegetation because this vegetation is 'azonal', which means that it is not climatically driven. Once the final potential vegetation map was generated, the distribution of aquatic vegetation was explicitly defined based on a soil map developed for Mexico at a scale 1:1,000,000 (INIFAP and CONABIO, 1995). At each point, we extracted values from 55 climatic variables obtained from the WorldClim site (Hijmans *et al.*, 2005). The dataset consists of 36 grids of monthly mean minimum temperature, maximum temperature and precipitation and a set of 19 bioclimatic variables.

The analysis was performed with 1000 trees. As part of its construction, random forests constructs successive independent trees using a bootstrap sample of the data set, each of which produces a vote (Breiman, 1996). In the end, the set of votes is used to generate a simple majority vote for prediction, or scores that provide basic probability estimates, which may then be used in weighted voting (Fawcett, 2006). We used majority vote prediction rules to generate the downscaled potential vegetation map. We validated the model using 256 inventory plots sampled near the transitions between different vegetation types (as these are the areas that are more inaccurate at the original scale). We used

Box 2.6 (cont.)

validation results to get insights into the performance acquired for the different vegetation classes and for overall model. Then we used receiver operating characteristics (ROC) curves to select the decision thresholds (cut-off thresholds based on probability estimates) that encompass the distribution limits for classes with low agreement and maximize overall prediction accuracy (Fawcett, 2006). At the end, we generated a potential vegetation map using the decision thresholds for these classes. Both the original and the climatically downscaled potential vegetation maps are shown in **Fig. 1**. The Kappa Index of Agreement showed an increase in accuracy from 0.40 for Rzedowski's map (95% confidence intervals between 32.5–48.7) to 0.80 for climatically-derived map (95% confidence intervals between 73.9–86.0). Overall accuracy increased from 55.5% to 85.9%. Estimated Kappa for each vegetation class was increased in all cases. Climatically-based random forests models can prove useful to increase the spatial resolution and accuracy of coarse-grained potential vegetation maps in mountainous areas with strong environmental gradients where important climatic variability is obscured at coarse-grained resolution. In conclusion, the proposed method is suitable to generate maps that can be appropriately used as a baseline for the estimation of the historical forest loss.

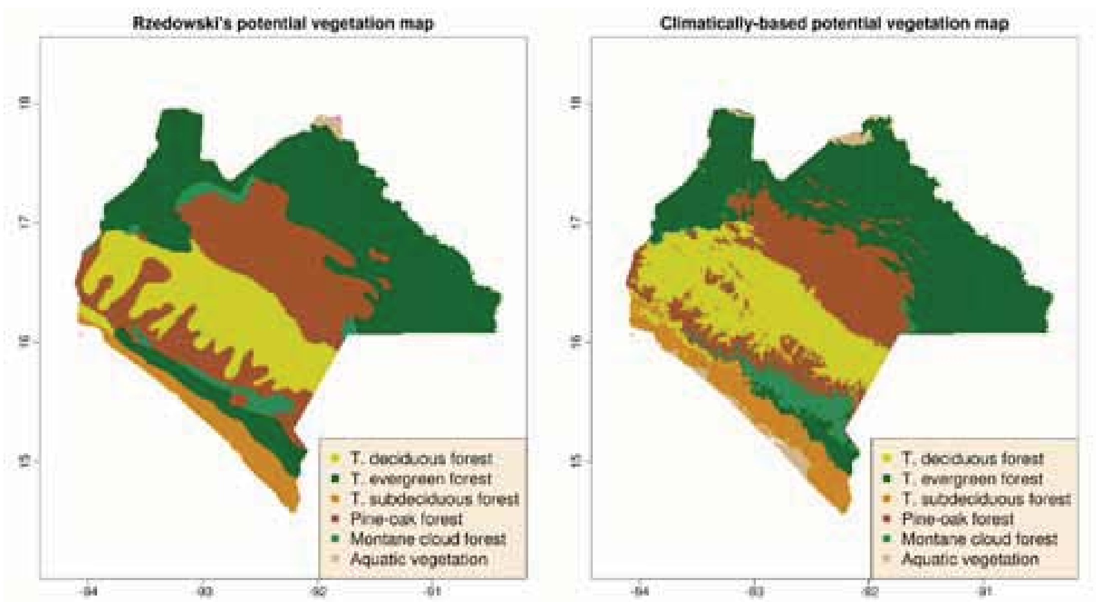


Figure 1 Rzedowski's potential vegetation map for Chiapas vs. climatically-based map developed with random forest. (T. = tropical).

production (Rundel, 1999; Balduzzi *et al*., 1982; Fuentes *et al*., 1986; Armesto *et al*., 2007). In addition, successional recovery of forest is usually constrained by continued pressure, water availability, soil erosion, lack of seed banks, disturbance by human-induced fires and limited regeneration capacities of forest species (Balduzzi *et al*., 1982; Fuentes *et al*., 1986; Conacher and Sala, 1998; Rundel, 1999; Armesto *et al*., 2007).

We found that loss of forest extent is partially mitigated by forest re-growth following the abandonment of farmland. In central Chile, we detected forest recovery on about 2.7% of the study area, a rate similar to that documented in other Mediterranean areas (Serra *et al.,* 2008). Forest recovery in central Veracruz may be partially explained as a by-product of comparing data from different types of satellites, but also by declines in agricultural area that may be linked to a reduction in subsidies and increased competition from the US following the NAFTA free-trade accords over the last decade (Pascual and Barbier, 2007). In Oaxaca, human migration may explain the decrease observed in cultivated land from 1989–1999 (INEGI, 2000). Deforestation in this region was concentrated in a few patches and land-use dynamics are apparently rapid, as traditional crop management involves a 5–10 year continuous cultivation period followed by a 10–15 year fallow period.

Drivers of forest change

We identified a number of biophysical and socioeconomic variables that were associated with changes in forest extent across our study areas (**Table 2.2**). Interestingly, the change in forest extent was explained by a unique combination of variables in each study area and the same variable may have either a positive or a negative effect in the different study areas, i.e. in different ecological, socioeconomical and cultural contexts (**Boxes 2.2 – 2.7**). For the entire time period addressed in this research, the biophysical variables with the strongest effects on change in forest extent were slope, insolation, and distance from remnant forest. Some of these variables drove forest change in opposite directions (loss or gain) in the different study areas. Thus, whereas the probability of an area experiencing forest loss was higher on gentle slopes, in accordance with our hypothesis, insolation showed different impacts in Chiapas (positive correlation) from central Chile and Oaxaca (negative correlation). Proximity to human settlements and farmland decreased the overall probability of deforestation in most study areas – an example of the 'curtain effect' – contrary to our hypothesis. Similarly human density was not found to have a major impact on deforestation. Distance from forests or roads had different effects in different study areas (**Table 2.3**), thereby partially confirming our hypotheses.

Table 2.3 Synthesis of results of selected multiple models for explaining the effects of various biophysical and socioeconomic variables on the conversion of forest to non-forest land-cover. The significance of the variables is represented by codes, being +++/--- = 0.0001, ++/-- = 0.001 and +/- = 0.01.

Variable	C. Chile	N. Argentina	C. Veracruz	Chiapas	Oaxaca
Elevation				-	+ + +
Slope	-	- - -	- - -	- - -	
Insolation	- - -			+ + +	- - -
Precipitation		- - -		-	
Distance to river		+ + +			
Distance to forest[1]	+ + +				- - -
Distance to settlements	+[2]	+ +[3]		+ + +[4]	+ + +
Human density					
Distance to roads	+ + +	- - -			+
Distance to agriculture	+ + +	+ +			+ + +
Distance to pasture			- -		

1 Distance within forest edge for the study area in central Chile
2 Cities >20,000 inhabitants
3 Towns <20,000 inhabitants
4 Access to towns >5,000 inhabitants

Box 2.7 Different sets of drivers across study regions

In central Chile, the multivariate logistic regression model for forest-no forest revealed that the probability of an area experiencing forest loss was highly significant (p <0.001) and positively related to the distance from the nearest forest edge for the four study periods, i.e. deforestation progresses from within forest fragments towards the edge, and produces treeless gaps within the forest. This variable had the strongest partial deviance of the four resulting models, accounting for at least half of the deviance explained by the final models achieved in the stepwise selection procedure. For the whole study period, the main explanatory variables of deforestation after distance from the nearest edge were distance from roads and distance from agriculture, all of which were positively correlated with the probability of deforestation, while insolation and slope were both negatively correlated and less relevant in terms of explained variance.

In northern Argentina, the probability of forest loss was highly related (p <0.001) to slope (negative correlation) and distance from rivers (positive correlation) in all study periods. Other significant explanatory variables were distance from roads and mean annual precipitation, showing that deforestation started close to roads and with relatively high precipitation (for dry forests), and moved away from those ideal conditions to current areas further away from roads and with lower precipitation. Distance from the agriculture frontier and from villages showed a significant positive effect, with higher deforestation rates in areas of higher accessibility and human presence.

In central Veracruz, univariate logistic regression models indicated that only four variables were negatively correlated with the probability of forest transformation. They included, in decreasing order of importance, slope, distance from pastures, distance from irrigation infrastructure, and aspect. When these same variables were incorporated into a multivariate logistic regression, the resulting model was highly significant according to the analysis of deviance, although the percentage of deviance explained was relatively low (19.4%). Slope and distance from pasture were found to be significant at the p <0.001 level, whereas the importance of distance from irrigation infrastructure (p = 0.063) and aspect (p = 0.09) declined.

In Chiapas, a GAM model which included elevation and annual rainfall explained the largest proportion of the deviance (12.2%). A model without rainfall explained 11.6% of the deviance, and slope became the variable most strongly associated with the probability that a pixel remained forested. After slope, radiation was the second most important variable both when taken alone and within a multivariate model. Access to large towns (>5000 inhabitants) had greater explanatory power as measured by deviance and partial deviance than access to small villages (>100 inhabitants). The probability that forest has been lost was found to be associated with high values of insolation during winter months, gentle slopes, accessibility to principal markets, low elevations, and low annual precipitation.

In Oaxaca, the probability of forest loss was highly significant and positively related to distance from crop fields and distance from villages in the four study periods. For the entire period (1979–2005), the main explanatory variables for deforestation with significant positive correlations were elevation and distance from agriculture, from villages and from forest; those with significant negative correlation were insolation and distance from forest.

Dryland forest areas in southern Argentina are undergoing a small but significant change to plantations with exotic conifers. Afforestation with exotic pines during the 1973–2003 period tended to occur at significantly (p <0.05) shorter distances from roads, urban areas and towns >1000 inhabitants. Distance from rivers or lakes seems to have been a poor predictor of afforestation. Contrary to expectation, it tended to occur at longer distances from small towns. Afforestation tended to occur at low elevations and on gentle slopes and at areas with mean annual precipitation higher than the average of the study area. This model was useful to detect possible land-use conflicts with the afforestation process. For example, it has been suggested that afforestation with exotic conifers has taken place at the expense of land potentially suitable for passive or active restoration of dryland forests of the native conifer *Austrocedrus chilensis*. If we compare potential for transition towards afforestation with habitat suitability for this species, we obtain the land-use conflict map presented in **Fig. 2.4.**

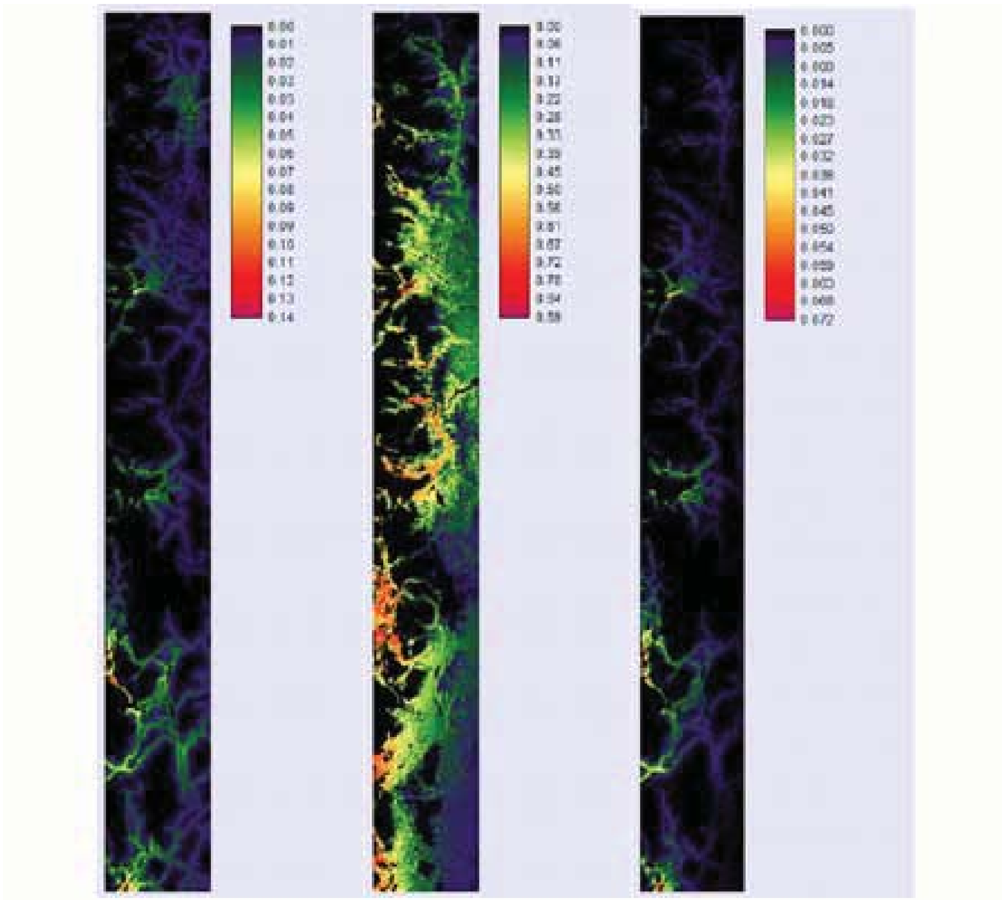


Figure 2.4 Application of the multivariate logistic model of afforestation to identify areas of potential dryland forest restoration in conflict with other land-use types in southern Argentina. *Left*: Map illustrating potential for afforestation with exotic pine species derived from this study. *Centre*: Habitat suitability model for *Austrocedrus chilensis*, the main native dryland forest species in the region. *Right*: Areas of conflict that are highly suitable for restoration of native dryland forest (or already have dryland forest) but are at risk of being converted to exotic pine plantations. Higher values indicate higher risk of conversion.

The differentiated analysis of drivers of deforestation indicated that the trend towards a reduction in natural vegetation cover was determined by a variety of biophysical and socio-economic contexts that resulted in different patterns of land-use types. For the entire time period addressed in this research, the biophysical variables with strongest effects on change in forest extent were slope (higher deforestation in flat areas), insolation, and distance from forest remnants. Slope and associated topographical barriers are fairly typical drivers identified in studies of tropical forest transformation (Geist and Lambin, 2002). In central Chiapas, fire is used to clear forests and to prevent woody re-growth in cattle pastures; therefore, slopes that receive more insolation in the dry season are more likely to be permanently cleared of woody vegetation than shaded slopes. Moreover, Echeverría *et al.* (Chapter 3) have shown that fire frequency and extent are increasing in these forest landscapes, making them more vulnerable to desertification.

Climate change, particularly rainfall patterns, may be also linked to deforestation (Grau *et al.*, 2005). In recent years, there has been an increase in rainfall in northwestern Argentina (Villalba, 1995), leading to expansion of the agricultural frontier and contributing significantly to the rapid increase in deforestation in the region. This has been accompanied by technological improvements (e.g. genetically modified soya) and high international demand that have raised product prices.

In central Chile, forest loss occurred with higher probability inside forest stands than at the border. As a consequence, our analyses also detected a higher probability of deforestation at larger distances from roads and agricultural fields. The same pattern has been observed in other studies (Ochoa-Gaona and González-Espinosa, 2000) and reveals hidden pressures from cattle grazing and illegal logging activities such as firewood collection and charcoal production (Armesto *et al.,* 2007). Such hidden pressures are not rare in Latin American countries (Callieri, 1996; Aubad *et al.*, 2008) where the rural population often depends on firewood for household consumption as well as the illegal production of charcoal for income generation. In central Veracruz, clearly, the rapid expansion of irrigated agriculture and cattle ranching increased pressure on native forests. Human activity-related factors affect forest fragment accessibility, as has been reported in other studies (Fujisaka *et al.,* 1996; Wassenaar *et al.,* 2007). Settlers converted land to pasture not only to raise cattle, but also to establish unpaved roads and collect firewood. After the forest fragments near the pastures are degraded, land conversion to agriculture or other pasture is more probable. In southern Argentina, deforestation is presumably produced by the occurrence of natural and anthropogenic fires, which in many cases do not regenerate back into forests and remain as stable grasslands or shrublands (Mermoz *et al.*, 2005). However, dryland forest areas in this region are undergoing a small but steady change to plantations with exotic conifers. This trend, in concert with other threats such as anthropogenic fires, livestock grazing and the introduction of exotic herbivores such as hares and rabbits are factors that hinder the restoration of dryland forest, which has been fragmented over centuries by native populations and European colonists (Veblen and Lorenz, 1988). Models of land-use/land-cover change can aid the identification of target areas of low conflict for a more rational planning of restoration efforts.

Implications for landscape planning and management

Human interactions with ecosystems are inherently dynamic and complex, and any categorization of these is an oversimplification. However, there is little hope of understanding these interactions without such simplifications (Ellis and Ramankutty, 2008). Working in multiple regions within Latin America enabled us to identify general trends at the regional scale that might be useful for landscape planning and serve as a basis for analyzing proximate drivers of land-cover change. However, the disadvantage of this approach is that it is more difficult to identify and assess patterns and process of land-use change at local scales in the real world (e.g. an individual field). Nevertheless, informal interviews that were conducted alongside the field surveys to establish classifications or to ascertain accurate assessments in most study areas provided an important complementary source of information to interpret the detected changes at the regional scale.

Natural vegetation loss and degradation reduce precipitation infiltration and runoff regulation, which promotes soil erosion, landslides and avalanches, and has a negative impact on ground water recharge (Conacher and Sala, 1998; Millennium Ecosystem Assessment, 2005b). In addition, vegetation cover is tightly associated with water balances within watersheds, biodiversity conservation, and regional climate regulation (Maass *et al.*, 2005; Feddema *et al.*, 2005; Foley *et al.,* 2005; Pielke, 2005). Land-use decisions therefore have consequences for the structure and function of ecosystems and affect provision of environmental goods and services; these decisions also affect humans in ways that go beyond the immediate land-use situation (Turner *et al.*, 2007). The continuous degradation of vegetation cover could have a strong impact on human livelihoods and well-being in the studied dryland landscapes, as there are increasing water demands for agriculture (Cai *et al.*, 2008) and human consumption owing to large population increases.

Environmental problems such as degradation, loss of biodiversity and decreases in productivity accumulate over the long term and have non-linear effects at regional to global scales (DeFries *et al.*, 2004; Foley *et al.*, 2005). Consequently, strategies for adapted land use, including the optimization of the spatial configuration of uses and restoration of the natural vegetation cover in critical areas should be developed quickly. Strategies should go beyond preservation within protected areas and logging restrictions along rivers and streams (Turner *et al.*, 2007). For instance, Rey Benayas *et al.* (2008) proposed the 'woodland-islet in agricultural seas' model to conciliate agricultural production and conservation or restoration of native woodlands. Closer monitoring is needed of livestock to establish guidelines for an adapted carrying capacity, as cattle also graze in forests. The repercussions of unsustainable firewood extraction and charcoal production have hardly been quantified in many regions, but we know that they impact strongly on forest conservation (see Chapter 6).

Land-use planning at the regional scale provides a unique opportunity for the establishment of general strategies that may, on one hand, accept or even promote deforestation at particular selected areas, and on the other hand maintain large forested areas suitable for sustainable timber and non-timber forest uses, and probably to a lesser extent, areas for conservation purposes. In northern Argentina, a land-use planning policy has been implemented over 10 million hectares of dry forests, zoning different land uses, from deforestation to conservation. Most of the forest corresponds to an intermediate category, theoretically oriented towards forest uses compatible with its own maintenance in the long run. In practice, most forest is heavily degraded and major efforts should be made to find economic incentives for the local inhabitants to reverse the degradation process and provide value to the remaining forests (see Chapter 10).

Apart from the need for land-use planning, restoration and rehabilitation are important issues in drylands (Le Houerou, 2000; Vallejo *et al.,* 2006). Long-term land-use intensification may represent unique cultural challenges for restoration efforts owing to the long history of human activity, the period of time during which dryland forest has been reduced and degraded, and generations of inhabitants have grown accustomed to its absence in the studied regions (Piegay *et al.*, 2005; Hobbs, 2009). In Chile, Holmgren and Scheffer (2001) postulated that there might be a window of opportunity for passive restoration through the exclusion of herbivores in El Niño Southern Oscillation (ENSO) years owing to higher water availability; this strategy could also be applied in southern Argentina given similar ENSO-climate connections. It could be especially interesting to use this strategy to establish buffer zones and corridors between remaining old growth forest, which were detected in this study as stable forest areas. Also, forms of adaptive and multifunctional land use such as mixed agroforestry systems should be encouraged as an alternative to monoculture cropping and crop pasture rotations (Ovalle *et al.*, 1996; Aronson *et al.*, 1998).

Conclusion

The research described here has provided quantitative estimates of forest extent and characterized the changes in land-cover in a wide variety of dryland landscapes under contrasting ecological, socioeconomic, and cultural scenarios. In addition, research examined the dynamics and drivers of forest loss that has taken place over the last ca. 30 years. We concluded that land-use intensification and limited natural regeneration continue to threaten dryland forest cover in many regions of Latin America, but that deforestation rates have diminished in the recent past compared to trends in the early part of the twentieth century, in accordance with global trends. The probability of an area experiencing forest loss was found to be higher on gentle slopes, and surprisingly, proximity to human settlements and farmland decreased the probability of deforestation in most study areas. Such analyses can help identify those areas that supported native forest in the past, and might therefore be considered as candidates for restoration. In addition, analysis of the factors responsible for forest loss and degradation can inform the development of restoration strategies and plans, by identifying those threatening processes that need to be addressed if restoration actions are to be successful.

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3 ASSESSING FRAGMENTATION AND DEGRADATION OF DRYLAND FOREST ECOSYSTEMS

C. Echeverría, T. Kitzberger, R. Rivera, R. Manson, R. Vaca, L. Cristóbal, G. Machuca, D. González, R. Fuentes

Introduction

Spatial patterns of forest cover can be understood as the spatial arrangement or configuration of forested ecosystems across a landscape (Forman and Godron, 1986). The importance of studying spatial patterns of forest cover is now widely appreciated, owing to the complex link between pattern and process in a landscape (Nagendra *et al.*, 2004), and the widely documented effects of habitat fragmentation on biodiversity. As a result, diverse studies have sought to develop measures of landscape pattern that may be used to monitor changes in forest cover (Sano *et al.,* 2009; Shuangcheng *et al.,* 2009; Zeng and Wu, 2005).

According to the driving factors that operate in a given landscape, spatial pattern can present a variety of different behaviours over time. For instance, loss and fragmentation of forest cover are among the most important transformations of landscape configuration occurring in many parts of the world (Carvalho *et al.*, 2009; Fialkowski and Bitner, 2008). On the other hand, pattern change associated with forest recovery or regeneration may lead to an increase of forest cover and connectivity (Baptista 2010; **Box 3.1**).

Box 3.1 Landscape features associated with the passive recovery of Mediterranean sclerophyllous woodlands of central Chile

A. Rivera-Hutinel, A. Miranda, T. Fuentes-Castillo, C. Smith-Ramirez, M. Holmgren

Although Mediterranean ecosystems are considered global hotspots of biodiversity and priority targets for conservation (Myers *et al*., 2000), they are among the most severely degraded and fragmented ecosystems in the world. In central Chile, land-cover of Mediterranean sclerophyllous woodlands (Chilean matorral) has been significantly reduced and transformed by a combination of human activities including logging, firewood extraction, vegetation burning, agriculture, livestock grazing, and the spread of exotic species of herbivores (Fuentes and Hajek, 1979; Holmgren, 2002; see Chapter 2). Ecosystems that have been highly degraded and extirpated from large areas, such as Chilean Mediterranean forests, are difficult and expensive to restore, especially because of the extremely dry and long summer period and strong impact of herbivory. Both factors, in addition to recurrent fire, can stop or retard successional processes (Fuentes *et al*., 1984). Frequently, severely degraded dryland ecosystems cannot be returned to their pre-disturbance condition without expensive management. The less costly strategies to restore vegetation cover in these ecosystems is to combine the passive regeneration of relatively less impacted areas, resulting from relatively slow natural processes, with active restoration activities that stimulate vegetation change from early successional stages to more mature and diverse forest.

Box 3.1 (cont.)

We assessed the regeneration potential of sclerophyllous woodlands of central Chile (33° S) over 50 years at three sites in the foothills of the Andes and two sites in the Coastal Range, and related the rates of vegetation change to specific landscapes features. Each study site (**Fig. 1**) was a mosaic of sclerophyllous forest and open pastures, with an average 40% of woodland cover and an extension of 700 ha on average (range: 631–911 ha) and had not been burned for at least two decades (1985–2008). Vegetation change was determined by comparison of aerial photographs taken in 1955 and 2007 over a regular grid of 250 m of points using standard supervised classification methods. We considered as evidence of woodland regeneration (1) a change in land-cover for a given point in the grid from bare soil or artificial grassland to forest cover. Persistence of the open cover condition was considered as a lack of forest regeneration (0). Any other observed changes in the vegetation or the maintenance of forest cover were excluded from the analyses. We related the recovery of forest cover to topographic variables (slope, orientation, altitude and exposure to solar radiation), as well as to spatial location of the regenerating patch (distance to the closest forest patch present in 1955, and distance to the nearest ravine). We used spatial regression models to control for spatial autocorrelation among sampling points.

We found an average rate of increase in land-cover of sclerophyllous forest from 0.4–1.0 ha/year. The probability of recovery of forest cover increased significantly at shorter distances from remnant (1955) forest patches, especially on south-facing slopes. This effect may be related to fact that patches can be a source of propagules but their environmental conditions may also facilitate tree seed germination and seedling survival (Fuentes *et al*., 1984, 1986; Holmgren *et al*., 2000). The spatial regression models also suggest that regeneration occurs in patches (at a 250 m scale), which could be related to local differences in grazing pressure, resource availability (nutrients and water) and micro-climatic conditions (temperature and air relative moisture).

Our work shows that Chilean sclerophyllous forest, which is considered strongly resistant to passive recovery from severe disturbance, can grow back in unburned sites under certain conditions. The proximity to existing forest patches or seed sources, slope aspect, and the aggregated patch structure of the vegetation are key features to be considered in the design of successful long-term restoration strategies to promote the passive restoration of Mediterranean sclerophyllous woodlands. The removal of herbivores, if possible, could accelerate the passive recovery of woodland vegetation cover (see also Chapter 8).

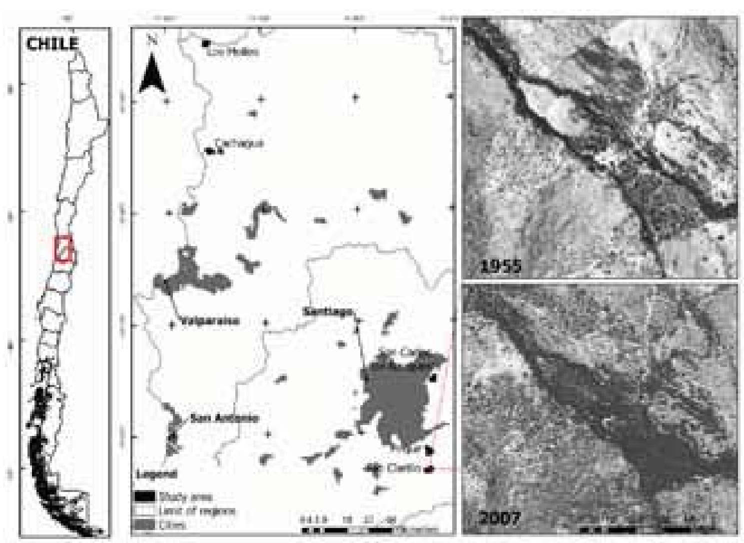

Figure 1 Location of the five study sites in central Chile (Cachagua, Los Molles, San Carlos de Apoquindo, Pirque). The aerial images show vegetation changes between 1955 and 2007 in Río Clarillo. Darker areas represent evergreen forest.

Central Valley of Chiapas, Mexico; tropical deciduous forest. Photo: R. Vaca.

Deforestation of seasonal dry premontane forest in northwestern Argentina. Photo: L. Malizia

Progressive deforestation typically results in an increase in the spatial heterogeneity, fragmentation, and edge characteristics of a forested landscape (Trani and Giles, 1999). In particular, fragmentation refers to the division of spatially continuous forest areas into isolated patches, which are separated by some other type of land-cover (such as agricultural land), commonly referred to as the landscape matrix (Forman and Godron, 1986). At the patch level, fragmentation causes an increase in patch isolation and edge, and a reduction of patch size (Echeverria *et al.*, 2006). In turn, this can increase the isolation of populations of individual species (Echeverría *et al.*, 2007), and can reduce population viability through its effects on key ecological processes such as dispersal, migration and gene flow (Giriraj *et al.*, 2010; Vergara and Armesto, 2009). As a result, forest fragmentation is now considered to be one of the principal causes of biodiversity loss (Baillie *et al.*, 2004). As forest loss takes place in a landscape, certain changes in the spatial configuration of the landscape can be observed (Cayuela *et al.*, 2006; Geri *et al.*, 2009). The analysis of the spatial attributes through landscape indices is a suitable approach to demonstrate the process of forest fragmentation at the landscape level (Zeng and Wu, 2005). Additionally, information on landscape structure can be used to inform forest management (Sano *et al.*, 2009).

Dryland systems are recognized as being of high biodiversity value, while representing the largest terrestrial biome on the planet (MEA, 2005; Schimel, 2010). Throughout the areas where they occur, dryland forests have been rapidly degrading and declining owing to anthropogenic disturbance (Hill *et al.*, 2008; Ravi *et al.*, 2010; Reynolds *et al.*, 2007). Loss of dryland forests has had a significant impact on carbon sequestration and temperature at the global scale (Rotenberg and Yakir, 2010). In Latin America, this ecosystem has been associated with human poverty, unhealthy living conditions and environmental degradation (Altieri and Masera, 1993). Management for the conservation and sustainable use of dryland forests should consider restoration approaches and habitat modification (McIntyre and Hobbs, 1999) with the aim of enhancing both biodiversity and human livelihoods. Few studies on the spatial pattern of dryland landscapes have been performed to examine the effects of human activity on dryland forests (Wang *et al.*, 2010), particularly in the context of their restoration.

In this chapter we present the results of research that assessed the trends in landscape patterns of forest cover in seven dryland study areas in Mexico, Argentina and Chile, by analyzing the dynamics of selected landscape metrics over the last four decades. Through a comparative analysis of these study areas, we identified both the high variability of landscape metrics and common trends in the spatial patterns of dryland forest. The aim of this research was to inform the development of plans for forest landscape restoration, one of the objectives of which is to restore connectivity in forest areas that have been fragmented (**Box 3.2**). Examination of the patterns and processes influencing forest fragmentation are therefore of direct relevance to the development of restoration approaches to be implemented at the landscape scale. The study landscapes included Veracruz, Oaxaca and Chiapas in México; Salta in northern Argentina and Bariloche in southern Argentina; and central Chile (**Fig 3.1**). A set of Landsat satellite images was classified and analyzed in each study area (see Chapter 2). Most of the study intervals spanned the past four decades, except for Chiapas where the period from 1990 to 2005 was analyzed.

Figure 3.1 Location of the study areas in Latin America.

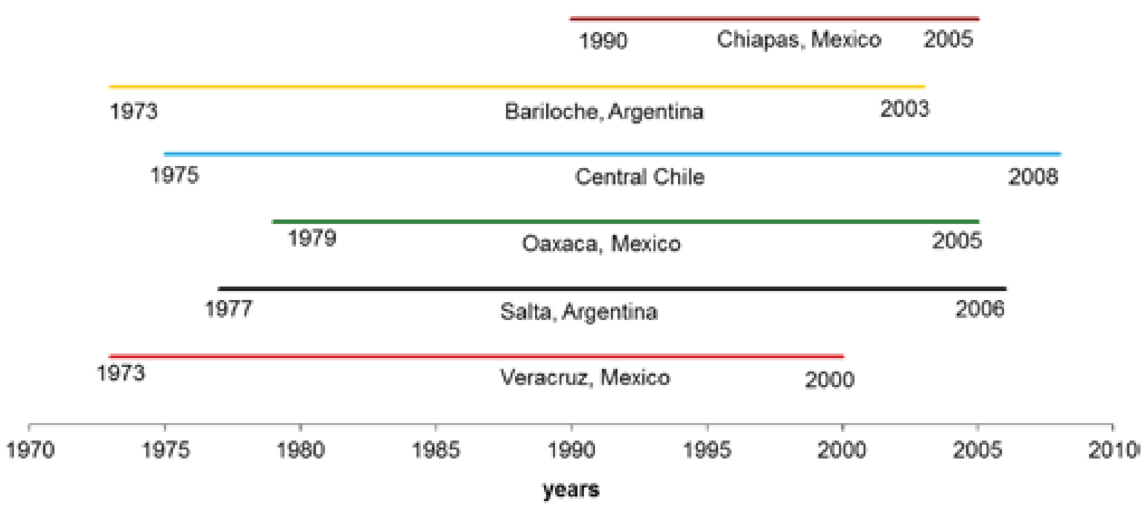

Figure 3.2 Study periods used for the fragmentation analysis in each study area.

Box 3.2 Landscape connectivity in the highly fragmented drylands of the Central Valley of Chiapas

R. Vaca, J.D. Golicher, L. Cayuela

The current pattern of forest cover observed in the Central Valley of Chiapas is the result of historic deforestation. We found that 68% of the original hypothesized area of dry forest (as defined by Olson *et al.*, 2001) was lost by 1990. The remaining forest (32% of the original putative area) is either retained by local landowners for its utility as a source of fuelwood and timber, or located in nature reserves or sites with steep slopes and low accessibility. Most of the forest in this region is highly fragmented, and only 19% is found in core areas, i.e. forest with a minimum distance of 110 m from the nearest patch edge. This landscape, dominated by human land use, presents a significant challenge for maintaining and conserving biodiversity.

Although most of the original forest cover had already been lost, the forest has not been completely cleared and replaced with an inhospitable matrix, as has occurred in other agricultural croplands or suburban environments of the world. The spatial concept of fragmentation in this case does not necessarily imply that habitat remnants are isolated by areas that function as hostile environments to the organisms within the remnants (Cayuela, 2009). The agricultural landscape still retains large isolated trees, woodlots, scattered groups of trees, secondary regrowth, hedgerows and living fences, amongst forest and shrubland patches of varying size, disturbance and management history. Together, they provide the habitats upon which the conservation of much of the flora and fauna in developed landscapes ultimately depends (Bennett, 1998; 2003). Even though regenerating or degraded forests and isolated trees may not provide all of the resources that a particular species need to survive, they may pose little resistance to the movement of many animals between patches and protected areas of forest where these resources are available (Bennett, 1998; 2003). In this context, an important priority for biodiversity conservation is to maintain a mosaic of semi-natural connected habitats within the agricultural land.

To investigate this issue, we measured dry forest connectivity in the study area (**Fig. 1a**) based on two different approaches, and we identified barriers to movement and priority actions for the region. The first approach focused on forest specialist species, i.e. species that have strict forest requirements. These species therefore require core areas for their survival over the long term. We focused on core areas of continuous forest cover larger than 5 ha. We developed a connectivity analysis based on distance matrices. Cores were considered neighbours if the smallest distance between their edges was less than 4 km **(Figure 1b**). The second approach focused on species that are less restricted in their forest habitat requirements, and can use isolated trees or small woodlots as well as core areas, and disperse easily through the matrix (e.g. some birds, insects, and many pioneer plant species). For these species, a highly fragmented landscape becomes more permeable to dispersion. In this case, we developed a connectivity analysis buffering away from any pixel classified as tree cover, using different buffer distances (100 m, 200 m, 300 m, 500 m, 1000 m, 5000 m, etc.). The map developed through this analysis shows the distance zones (proportion of area) between pixels classified as tree cover (**Figure 1c**). This analysis thus allowed the recognition of areas of decreasing permeability to movement.

Box 3.2 (cont.)

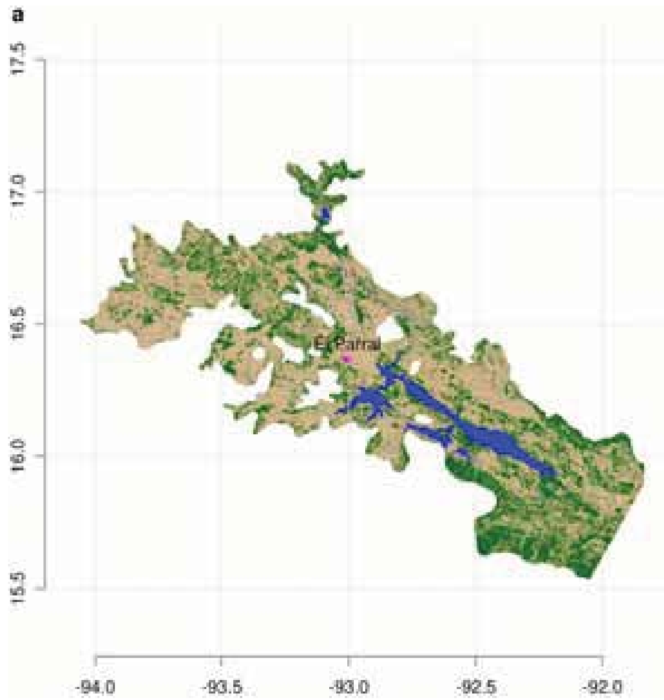

Figure 1a Study area, the Central Valley of Chiapas: forested areas are represented by dark green and the non-forested matrix by tan.

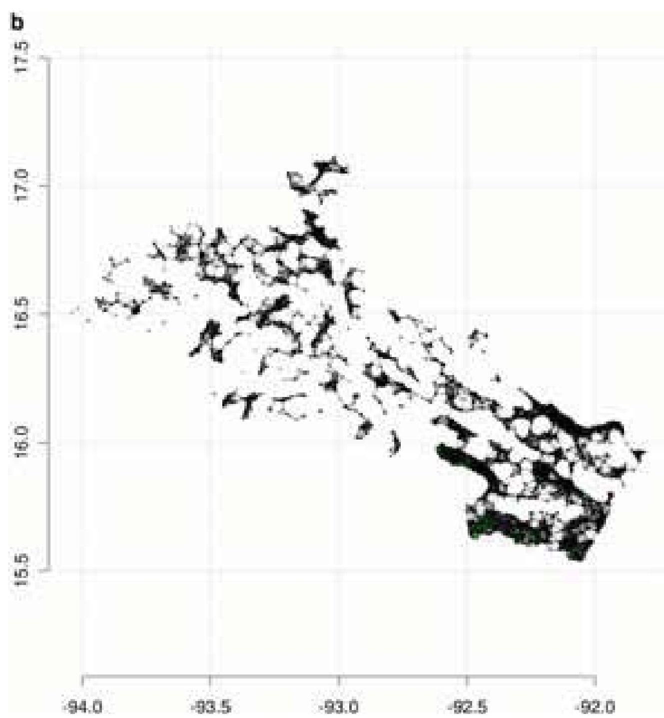

Figure 1b Connectivity analysis based on distance matrices: black lines represent Euclidean distances of less than 4 km between the edges of cores with areas higher than 5 ha.

Box 3.2 (cont.)

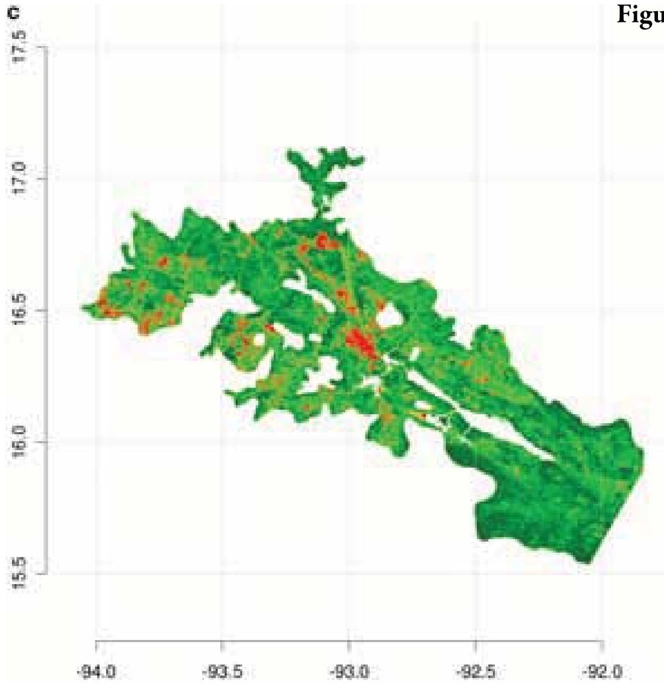

Figure 1c Connectivity analysis based on buffers of different distances, from any pixel classified as tree cover. Forested pixels are represented by dark green, and buffers of increasing distance are represented by a colour gradient: light green (buffer areas from 0 to 200 m), orange (buffer areas from 200 to 500 m), and red (buffer areas from 500 to 5000 m). This colour gradient represents the proportion of area between pixels classified as tree cover, pointing out areas of decreasing permeability to movement.

Barriers to movement tended to coincide spatially using both approaches. Results suggest that core areas were generally not well connected, especially in the centre of the study area. Nevertheless, isolated trees and small patches may enhance connectivity considerably for mobile organisms. The distance between any form of tree cover was generally below 200 m. The lowest connectivity was found in the area around El Parral (pointed out in **Fig. 1a**). But even in this area, trees were still present (**Fig. 2**). Biodiversity conservation can be achieved by maintaining the diffuse mosaic of forest, open woodland and scattered trees, but also through restoration of habitats focused both on linking core areas and increasing permeability. The forest can be conserved by working with landholders in order to minimize human impacts in the remaining forest patches, many of which are greatly disturbed and degraded as a result of livestock grazing. Future actions for increasing connectivity and permeability should target the restoration of degraded pastures, the development of fuelwood plantations, and the expansion of living fences, shade and forage trees within the landscape. Finally, further actions should be focused on protecting and managing major links between conservation reserves to assist their long term viability.

Figure 2 Deforested drylands near El Parral in the Central Valley of Chiapas, Mexico.

Cropland in dry forest areas of Chile. Photo: C. Echeverria

Dry forest in central Veracruz, Mexico. Photo: C. Alvarez

Techniques to quantify spatial patterns of forest cover

Analysis of forest fragmentation was conducted using the following set of selected landscape metrics: (a) patch area (ha), (b) proximity index, (c) patch density (n/100 ha), (d) total edge length (km), and (e) largest patch index (LPI, %). All of these metrics reflect the different effects of fragmentation on the spatial attribute of forest patches. Index proximity was calculated for a radius of 1 km and core area for an edge depth of 50 m. In addition, we estimated aggregation index and adjacency index between forest cover and the major land-cover types. It is expected that the aggregation of forest patches decreases as a result of fragmentation and increases with forest contiguity. Similarly, the adjacency between native forest and human-induced land-cover types should increase with changes in the matrix.

A minimum mapping unit of greater than 5 pixels was used for the spatial analyses. This enabled differences in data quality produced by the resampling of the MSS images to be minimized. Map preparation was performed using ARC MAP (version 3.3; ESRI, 2009). Landscape metrics were computed by FRAGSTATS (version 3) (Mcgarigal *et al.*, 2002) to compare the spatial patterns of forest cover for each time interval and study area.

Mapping spatial patterns of forest cover

Maps of forest cover based on patch size were generated for each study area and for each study year (**Fig. 3.3**). Most of these maps provide evidence of typical patterns of anthropogenic landscape change. The patterns are comparable to those observed in many other parts of the world (Abdullah and Nakagoshi, 2006; Wang *et al.*, 2010). Deforestation and fragmentation of dryland forests have occurred in most of the study areas, except in Bariloche where some forest fragments moved to upper classes of size during the study interval (**Fig. 3.3c**), and in Chiapas, where forest fragments did not appear to change in size over time (**Fig. 3.3b**). Maps of forest fragmentation showed a considerable increase in the number of smaller patches over time in Xalapa, Oaxaca, central Chile and Salta (**Figs. 3.3a, d, e** and **f** respectively).

Burned stand of *Austrocedrus chilensis* in Southern Argentina. Photo: J. Birch

Figure 3.3 Temporal variation in patch size for the different study areas: (a) central Chile, (b) Chiapas, Mexico, (c) Bariloche, Argentina, (d) Salta, Argentina, (e) Veracruz, Mexico, (f) Oaxaca, Mexico. Larger patch sizes in green, smaller patch sizes in red, intermediate patch sizes in orange.

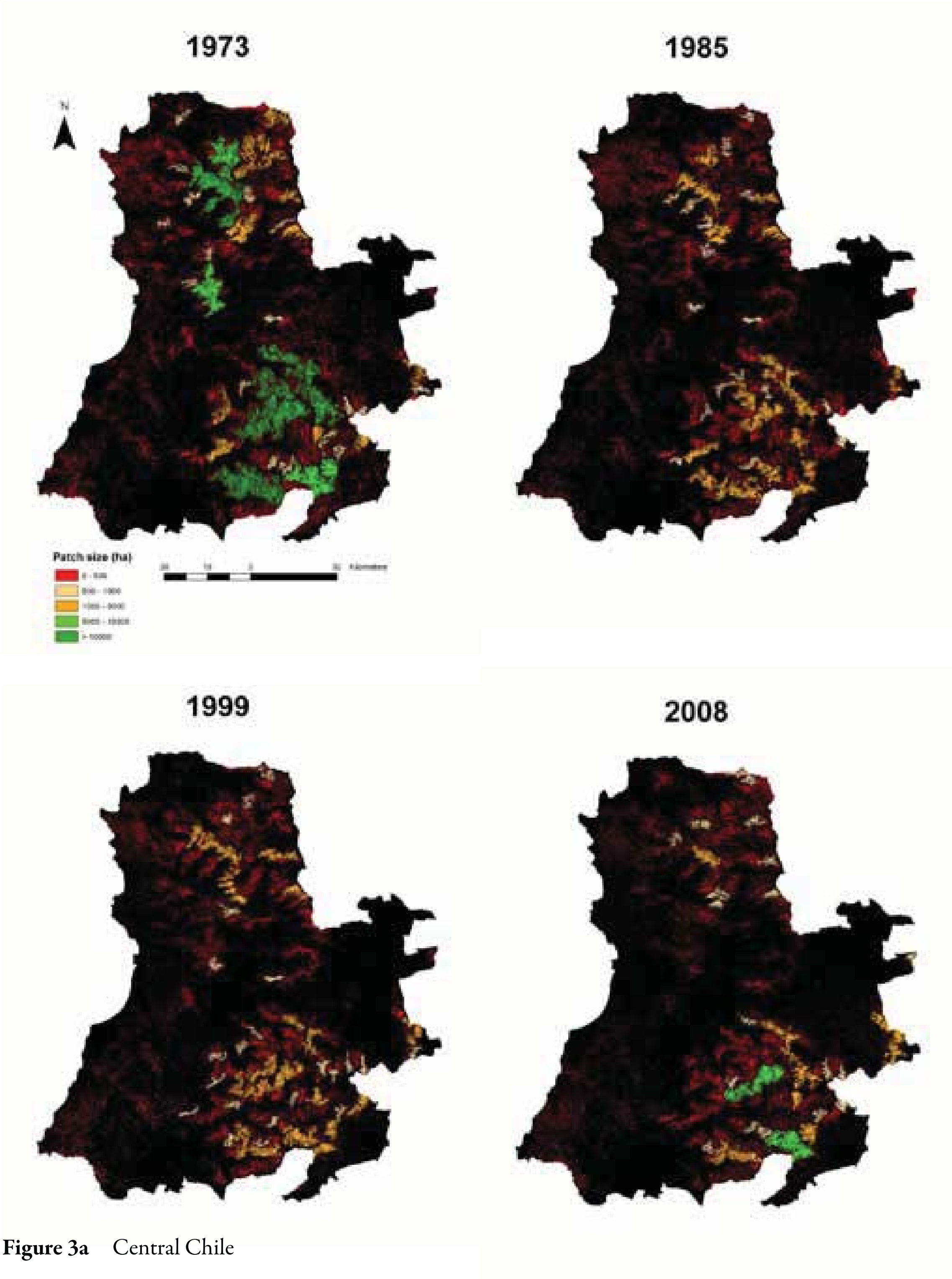

Figure 3a Central Chile

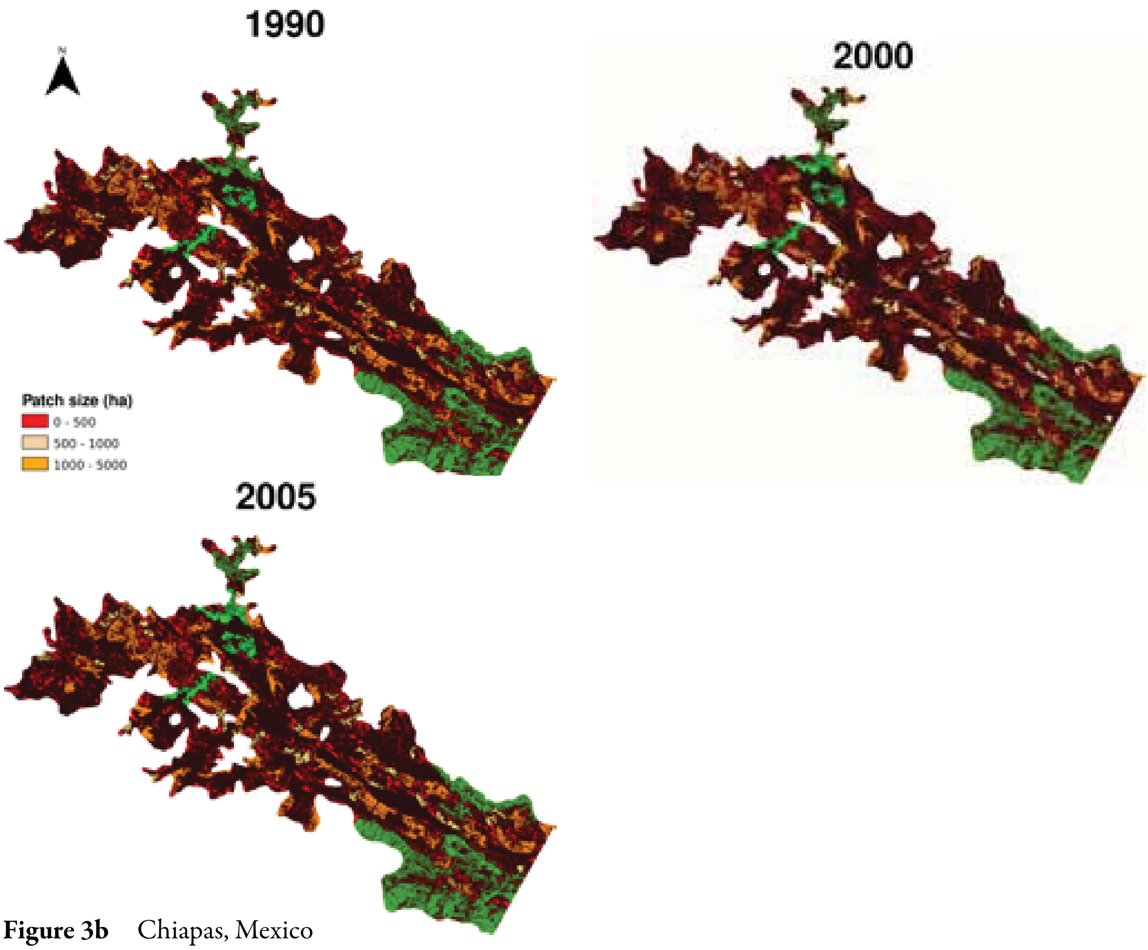

Figure 3b Chiapas, Mexico

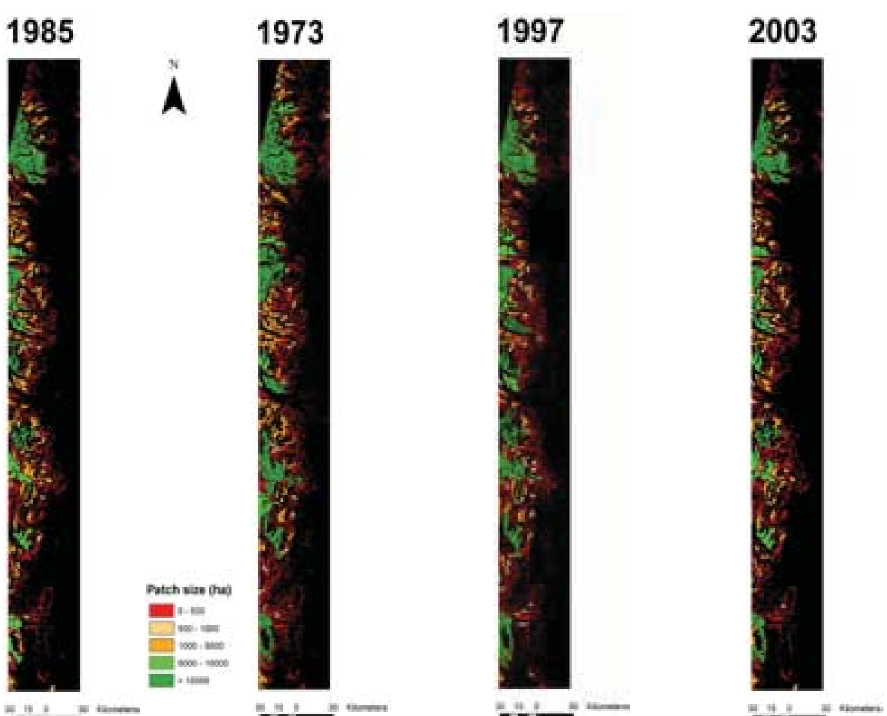

Figure 3c Bariloche, Argentina

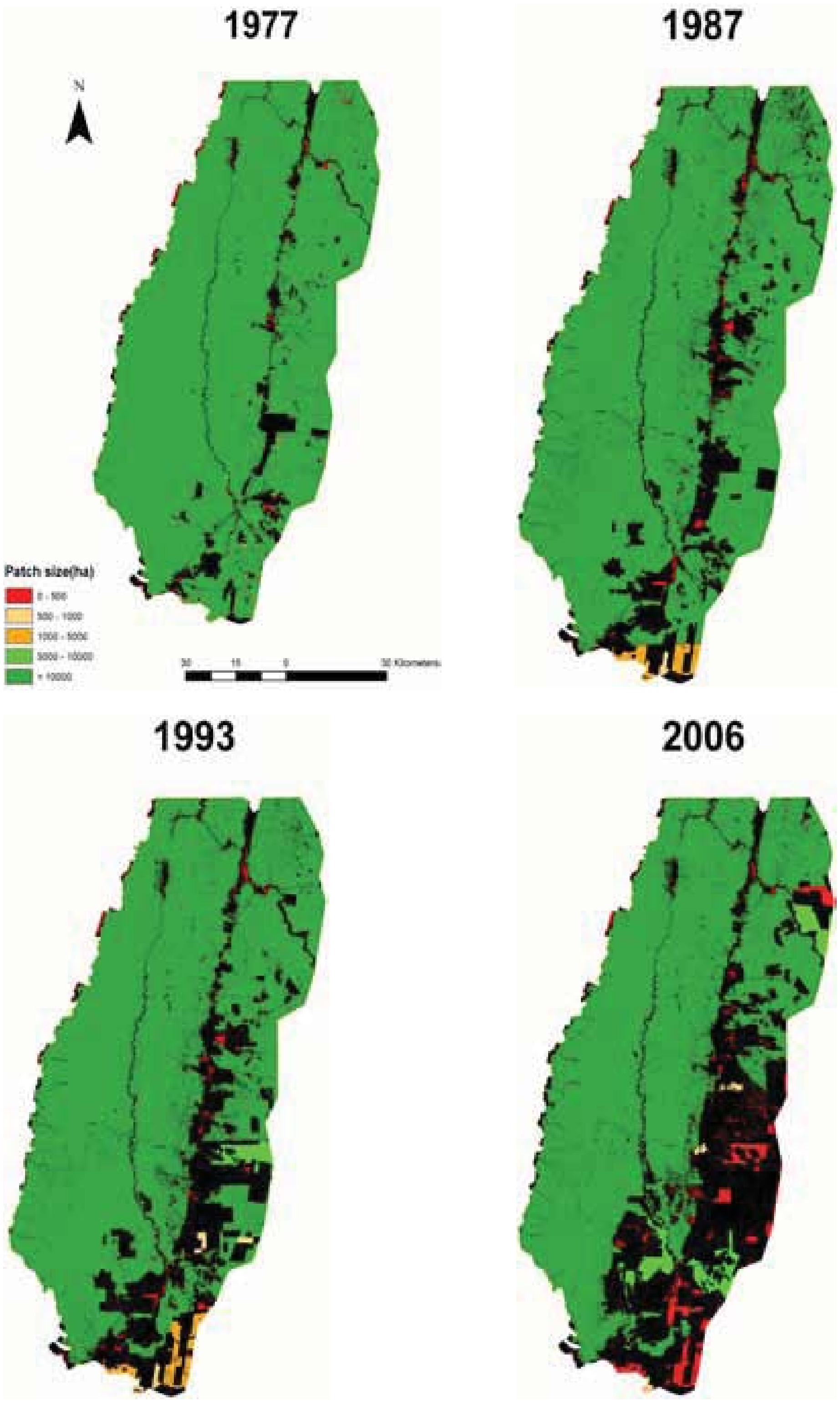

Figure 3d Salta, Argentina

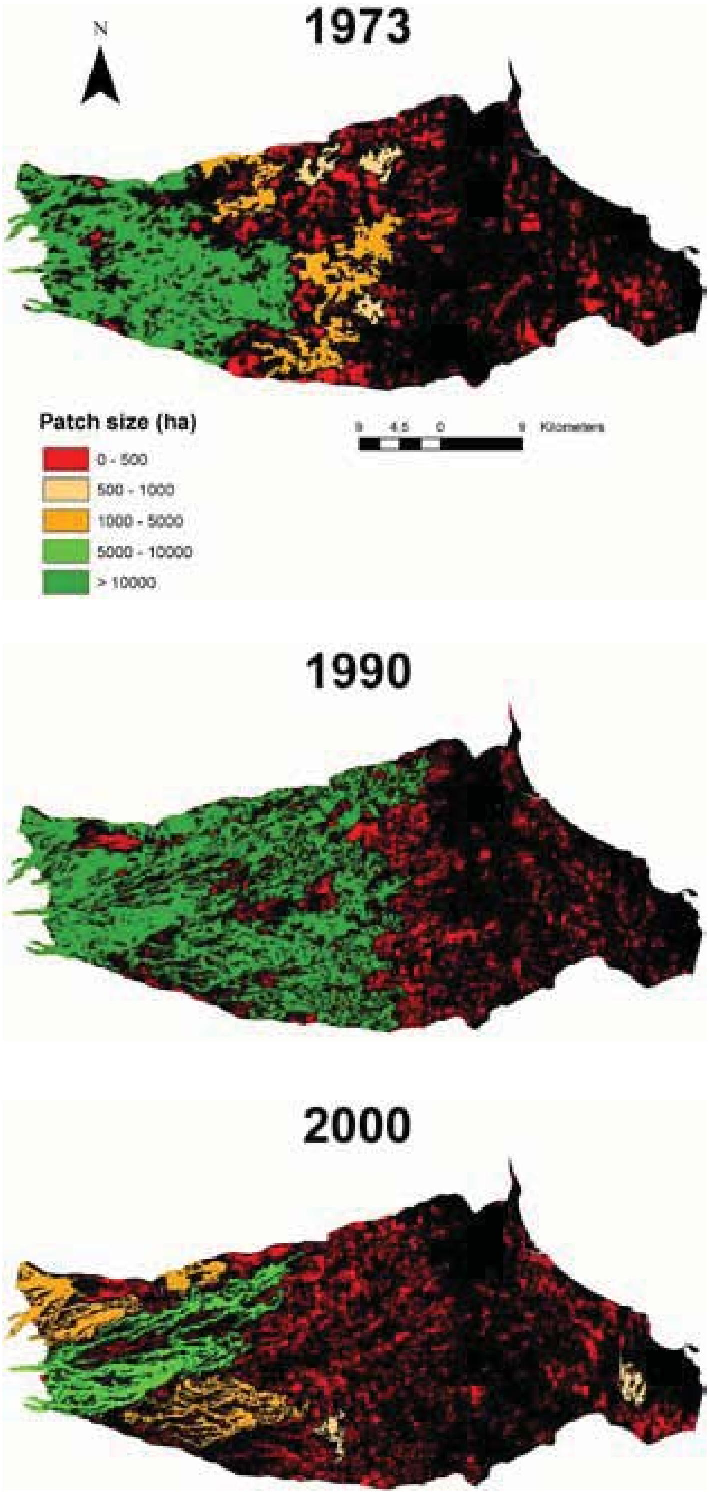

Figure 3e Veracruz, Mexico

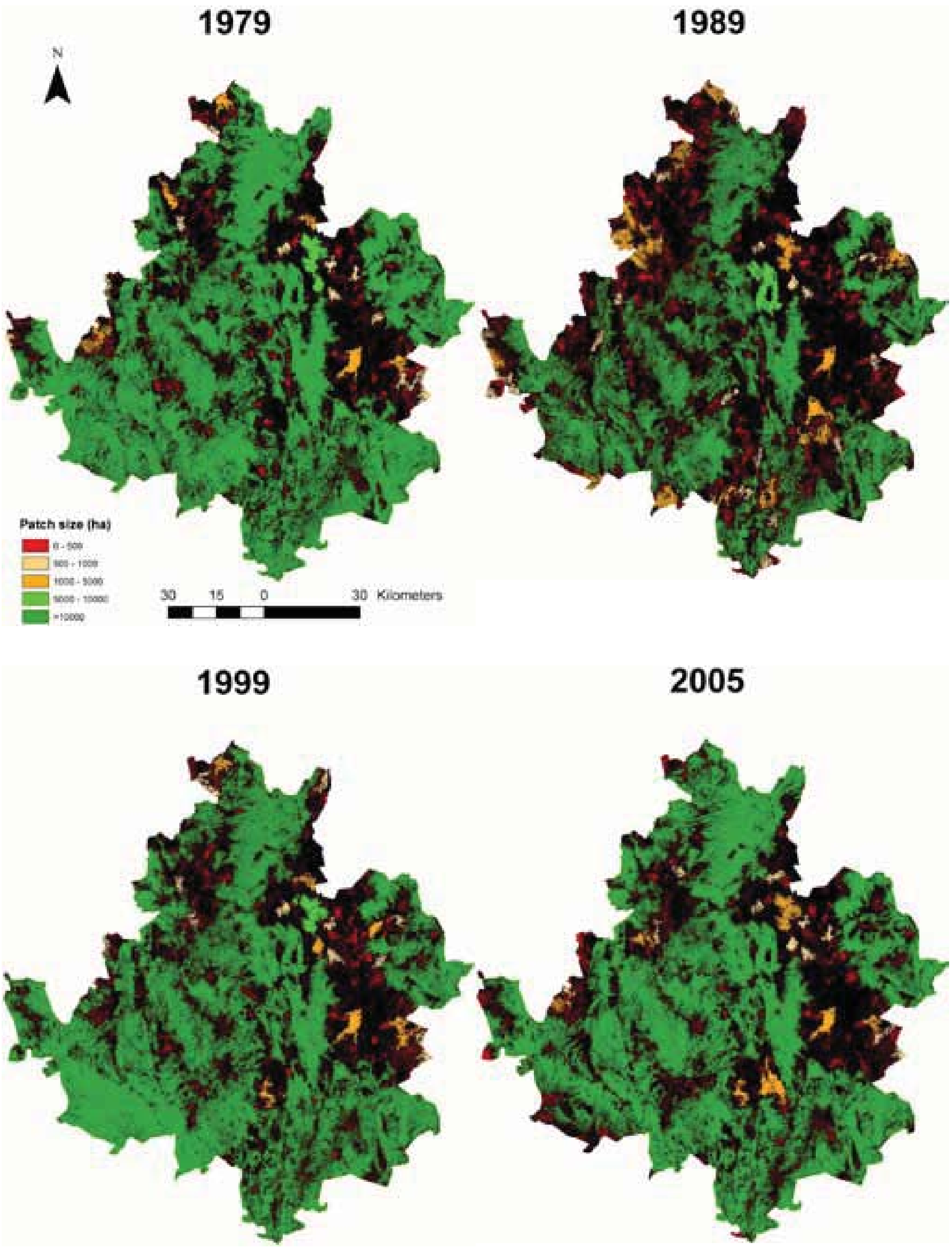

Figure 3f Oaxaca, Mexico

Analysis of spatial patterns of dryland forests

Most of the study areas exhibited a decline in patch size between the earliest and the most recent maps. In particular, Veracruz, Oaxaca, central Chile and Salta showed a decline in patch size, while Chiapas remained practically constant and Bariloche exhibited an increase in this variable (**Table 3.1**). In Oaxaca and central Chile, the total edge length of forest patches increased and then declined over time (**Table 3.1**). In Veracruz and Salta there was an increase in the total of edges. In contrast, Bariloche was the only study area that exhibited a permanent decline in the number of fragment edges, whereas Chiapas did not demonstrate changes in this variable over time. With respect to the core areas of forest fragments, all of the study areas exhibited a decline in this index through time, except for Bariloche, which showed an increase (**Table 3.1**). The greatest declines occurred in central Chile and in Veracruz, where 66% and 51% of the core area was lost during the study periods respectively. In contrast, Chiapas did not present a substantial change in this index (1.3%), while in Bariloche the core area increased 16% (**Table 3.1**). Index of proximity (which provides a measure of the degree of isolation) decreased in Chiapas, central Chile and Salta (**Table 3.1**). In Veracruz and Oaxaca, this index varied during the study period, without providing a clear trend. In Bariloche this index presented an increase between 1973 and 1997 and then it declined during the last time interval.

These trends in landscape indices were associated with variation in patch density (**Fig. 3.4**). Owing to the fact that the number of patches may increase as a result of the creation of new patches by fragmentation, a further decline can be observed either by the loss of the new forest patches or the union of patches as a result of forest regeneration. This trend enabled different stages in the spatial dynamics of forest to be identified. In Veracruz and Salta, a gradual increment in patch density (**Fig. 3.4**) and edge length, and a decline in patch size and core areas (**Table 3.1**) characterized a landscape affected by progressive fragmentation during the study periods.

Oaxaca is the only study area where the patch density and edge length were curvilinear, with metrics changing direction at the half-way point of the study period (**Fig. 3.4**). This reflects a rapid division of forest patches that were later eliminated by high rates of deforestation. In central Chile, the number of patches gradually decreased owing to the conversion of forest patches (**Fig. 3.4**). This pattern was associated with a decrease in core area and an increase in patch isolation, and with a continuous loss of forest fragments over time (**Table 3.1**). In contrast to this situation, in Bariloche the increase in patch size, core area and proximity index and a decline in patch density and edge length showed that the forest cover was recovering, showing an opposite trend to forest fragmentation (**Fig. 3.4** and **Table 3.1**). In Chiapas, the slight decline in patch density (**Fig. 3.4**), and the almost constant values of patch size, edge length and core area, revealed a low level of forest fragmentation in this landscape and the stabilization of forest cover (**Table 3.1**).

During the study periods, the largest forest patch occupied no more than 2% of the whole landscape in each of Chiapas, central Chile and Bariloche. On the other hand, in Salta this index reached a higher value, ranging from 52% to 32% between 1977 and 2006. In Oaxaca values varied slightly from 24% to 23%, and in Veracruz, from 6% to 2%.

Table 3.1 Landscape metrics estimated for the six study areas.

Veracruz, Mexico

Landscape indices	1973	1990	1999
Mean patch size (ha)	139.7	73.9	27.9
Total edge length	3,091,320	6,334,110	6,345,420
Total core area (ha)	44,160.84	41,404,14	21,771.7
Mean proximity	1,235.1	16,712.5	1,229.35

Oaxaca, Mexico

Landscape indices	1979	1989	1999	2005
Mean patch size (ha)	99.9	22.9	41.2	46.7
Total edge length	64,069.8	105,900.9	106,688.9	89,516.0
Total core area (ha)	514,323.3	2,246,893.7	386,258.9	428,649.66
Mean proximity	332,727.6	28,168.4	366,210.5	388,341.7

Chiapas, Mexico

Landscape indices	1990	2000	2005
Mean patch size (ha)	13.1	14.1	14.4
Total edge length	113,509.0	110,960.3	111,196.6
Total core area (ha)	277,821.0	275,821.0	274,287.0
Mean proximity	4,432.03	4,174.92	4,164.15

Central Chile

Landscape indices	1975	1985	1999	2008
Mean patch size (ha)	8.8	6.3	6.2	6.0
Total edge length	44,400.1	49,837.8	50,768.7	41,897.8
Total core area (ha)	76,901.2	29,922.8	23,500.2	26,149.2
Mean proximity	1,028.0	454.9	380.9	427.1

Salta, Argentina

Landscape indices	1977	1987	1993	2006
Mean patch size (ha)	1,074.7	757.9	528.6	330.2
Total edge length	8,540.1	15,826.2	15,455.0	14,872.7
Total core area (ha)	682,693.0	614,457.0	581,090.0	506,464.0
Mean proximity	299,131.0	522,803.0	217,447.0	174,601.0

Bariloche, Argentina

Landscape indices	1973	1985	1997	2003
Mean patch size (ha)	8.85	14.3	12.2	13.9
Total edge length	79,359.68	52,586.80	62,149.86	52,021.08
Total core area (ha)	115,080	135,654	145,918	132,901
Mean proximity	1854	2226	2336	2059

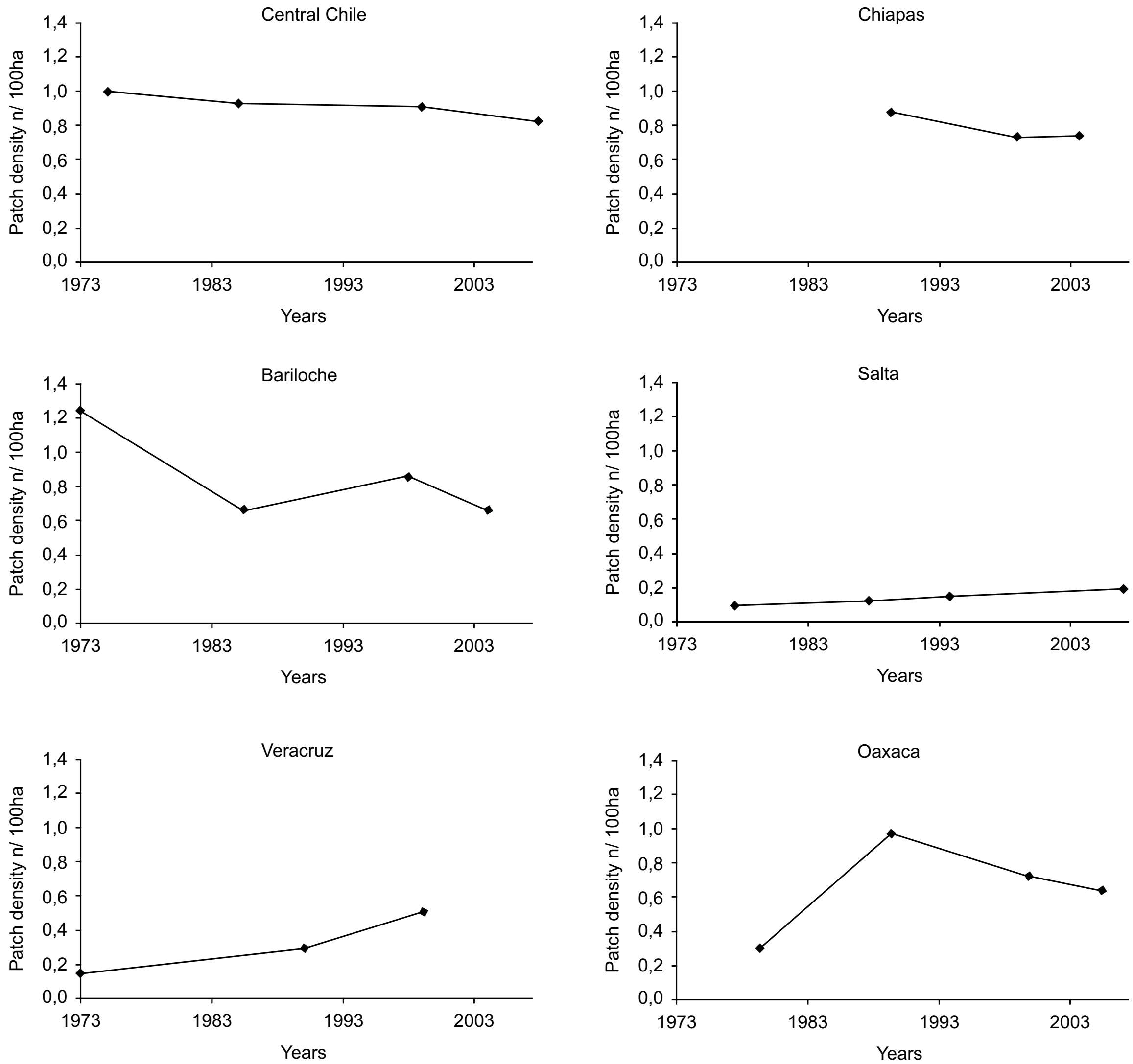

Figure 3.4 Temporal variation in patch density in the six study areas: (a) central Chile, (b) Chiapas, Mexico, (c) Bariloche, Argentina, (d) Salta, Argentina, (e) Veracruz, Mexico, (f) Oaxaca, Mexico.

As forest loss continues, it is expected that the largest patch index (LPI) will decline owing to a division of large patches by fragmentation (Trani and Giles, 1999). By graphing the forest loss versus LPI for the study areas, it was observed that in Oaxaca, Veracruz and Salta (**Fig. 3.5**) a continuous fragmentation has led to a division of large forest patches, causing a decline in the LPI. However, in central Chile and Bariloche, there was a slight increase in LPI as forest loss increased (**Fig. 3.6**). This opposite trend was the result of the union of large forest fragments despite the loss of others (**Fig. 3.3**). In Chiapas the LPI did not show variation, owing to the fact that forest area remained almost constant during the study period (**Fig. 3.6**).

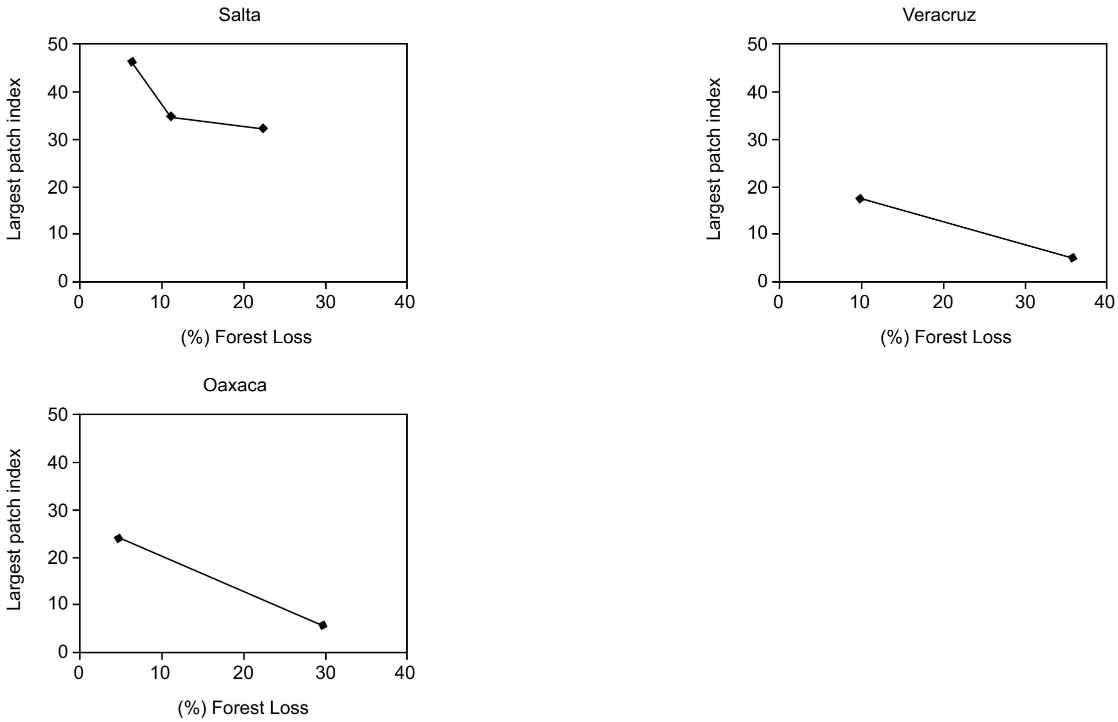

Figure 3.5 Relationships between forest loss and largest patch index in study areas where the largest patch represents more than 4% of the area of the landscape: (a) Salta, Argentina, (b) Veracruz, Mexico, (c) Oaxaca, Mexico.

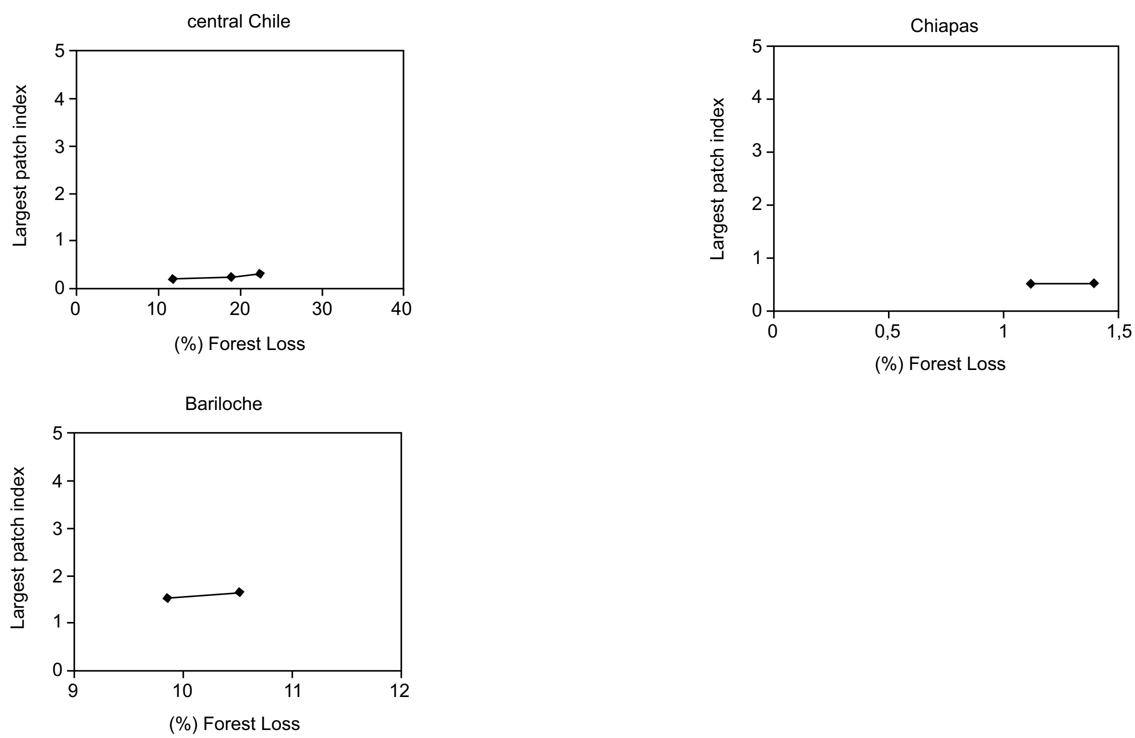

Figure 3.6 Relationship between forest loss and largest patch index in study areas where the largest patch represents less than 4% of the area of the landscape. Different scales in the x axis were used for clarity: (a) central Chile, (b) Chiapas, Mexico, (c) Bariloche, Argentina.

Results also showed different rates of forest disaggregation over time, illustrated by the degree of forest loss and fragmentation (**Fig. 3.7** and **3.8**). The greatest decline in forest aggregation was observed in central Chile, where this index decreased from 82% to 65% across the study period, most rapidly during the first time interval (**Fig. 3.7**). However, in the earliest study years, Salta and Veracruz exhibited the highest levels of forest aggregation or spatial integrity, with 99% and 96% respectively (**Fig. 3.7** and **3.8**). In central Chile the disaggregation of forest cover was accompanied by a loss of forest patches rather than by a division of forest patches, as demonstrated by values of patch density (**Fig. 3.4**). On the other hand, in Veracruz and Salta the number of patches increased (**Fig. 3.4**) while patch size declined (**Table 3.1**), reflecting a gradual decline in the level of forest aggregation or an increase in forest fragmentation. Chiapas remained constant with 85% forest aggregation, indicating no change in spatial patterns (**Fig. 3.7**).

Oaxaca was the only study area that exhibited a decline and further increase in the aggregation index over time (**Fig. 3.12**). By comparing this result with the values generated for the other metrics, it can be observed that during the first time interval, forest cover was disaggregated by the division of a large patch, which resulted in an increase in the number of patches (**Fig. 3.8**). Between 1989 and 2005, the forest became more aggregated, increasing the patch size (**Table 3.1**). In Bariloche, forest cover showed a gradual increase in aggregation (**Fig. 3.12**), indicating the recovery of new patches and an increase in patch size (**Table 3.1**)

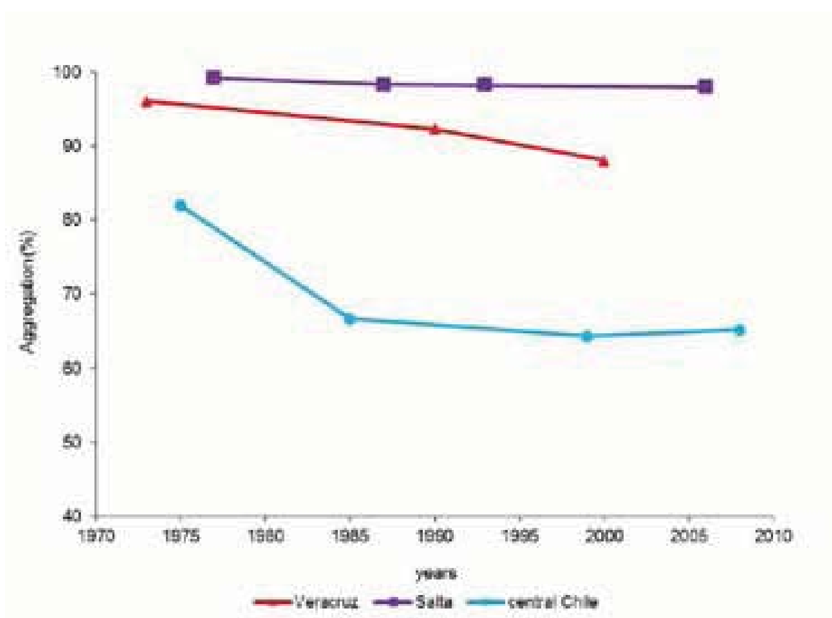

Figure 3.7 Aggregation index of forest cover in study areas where this index exhibited a decline (Veracruz, Salta and central Chile) or was constant (Chiapas).

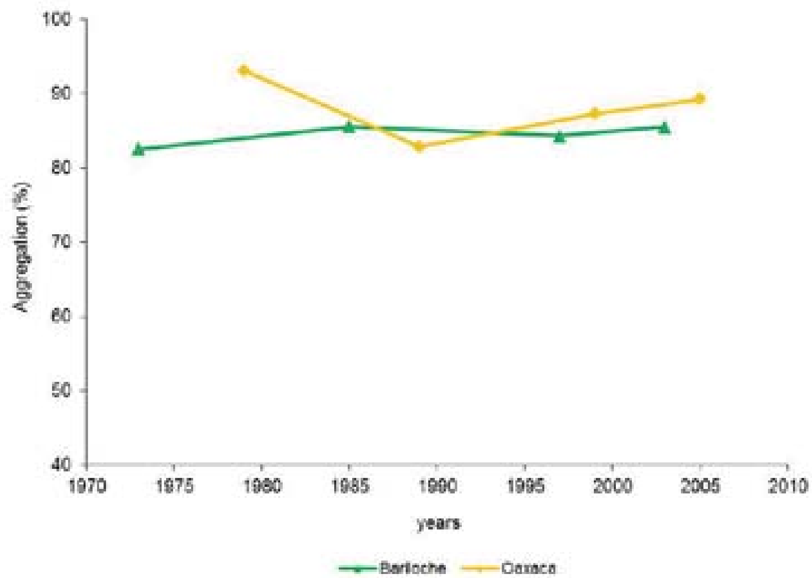

Figure 3.8 Aggregation index of forest cover in study areas where this index exhibited an increase (Bariloche and Oaxaca).

Changes in the spatial patterns of the dryland forests are explained largely by changes in neighbouring land-cover. Interestingly, in three study areas forest fragments were primarily surrounded by croplands while in the other three areas, the fragments were adjacent to degraded forest (in the case of Salta and central Chile) or to shrubland (in Bariloche) (**Figs. 3.13** and **3.14**). In most of the study areas the dryland forest fragments were surrounded by more than 70% croplands or by degraded forest and shrubland. This high percentage of adjacency to human-induced land uses indicates that most forest edges may be subjected to anthropogenic activities that could potentially affect the survival of many species.

Oaxaca and Veracruz showed greater dynamics in the percentage of adjacency between croplands and forest fragments (**Fig. 3.9**) than in study areas surrounded by degraded forest or shrubland (**Fig. 3.10**). The greater variation in Oaxaca and Veracruz is related to changes in the matrix composition represented by a replacement of grassland and bare ground by cropland, particularly during the 1970s and 1980s. Later in the 1990s, the adjacency to cropland tended to decline in Veracruz owing to the expansion and replacement of cropland by grassland. In Chiapas and in Oaxaca, more than 90% of forest fragments were by cropland during the last decade (**Fig. 3.9**). On the other hand, in all the South American study areas, forest patches were largely surrounded by degraded forest or shrubland (**Fig. 3.10**). In Salta a gradual decline in the adjacency between forest and degraded forest was observed after the mid–1980s. In central Chile, which recorded the greatest adjacency to degraded forest, a kind of 'pseudo-savannah' named *espinal* was reached by 2000, which then declined by 2008 (**Fig. 3.10**).This variation was related to the dynamics of the *espinal* that was converted to cropland, and which originated from the degradation of sclerophyllous forest.

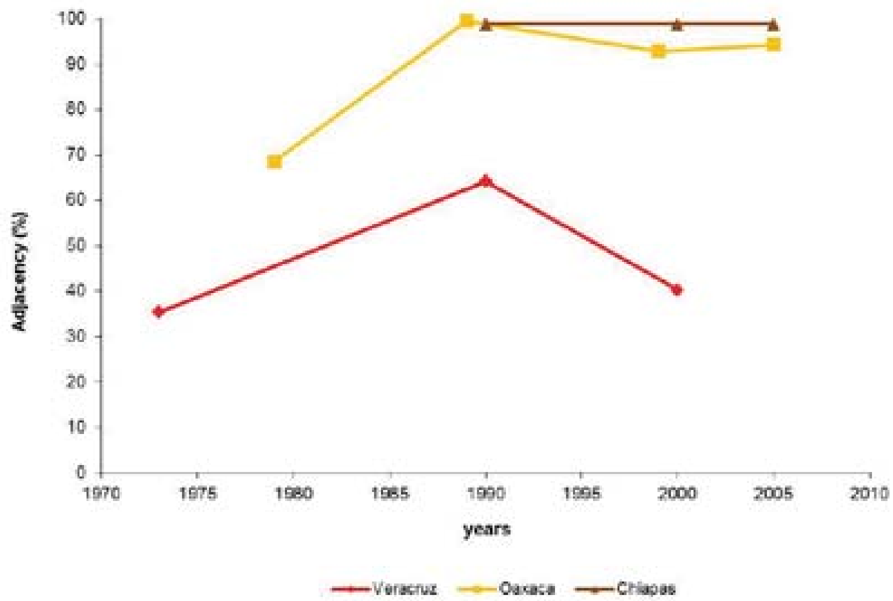

Figure 3.9 Percentage of adjacency between dryland forest patches and cropland in Chiapas, Oaxaca and Veracruz. Cropland was the major land-cover type adjacent to forest patches in these study areas.

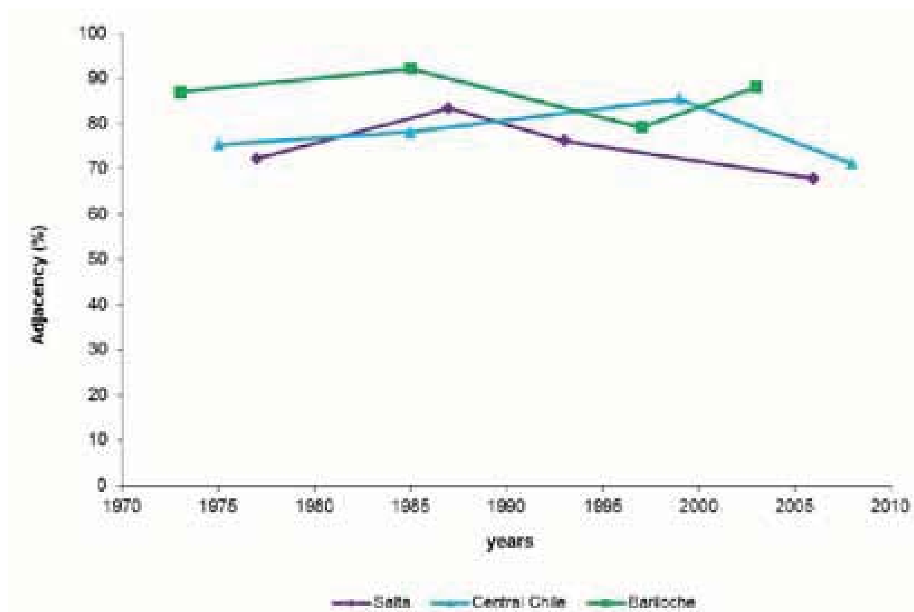

Figure 3.10 Percentage of adjacency between dryland forest patches and degraded forest/shrubland in Salta, Bariloche and central Chile. Degraded forest and shrubland were the major land-cover types adjacent to forest patches in these study areas.

Landscape states and habitat destruction

McIntyre and Hobbs (1999) discuss the process of habitat destruction and habitat modification, which can be conceptualized as a continuum, associated with the influence of human disturbance. They identified four types of landscape states along a gradient of destruction: intact (<10% habitat destroyed), variegated (10–40%), fragmented (40–90%) and relictual (>90%). In variegated landscapes, the habitat still forms the matrix, whereas in fragmented landscapes, the matrix is composed of destroyed habitat. During the study period, the study areas exhibited different degrees of forest loss (**Fig. 3.3** and **Table 3.2**). In Veracruz, the percentage of remaining forest cover (see **Table 2.1** in Chapter 2 on recent loss) was 9% in 2000, which corresponds to a relictual landscape. In this area the effect of intensive agricultural development has led to a progressive fragmentation of forest habitat and to a high rate of forest loss (**Fig. 3.5**).

On the other hand, the percentages of forest cover detected were 16% in Bariloche, 32% in Chiapas, 35% in central Chile and 60% in Oaxaca. These study areas correspond to fragmented landscapes characterized by more than 40% of habitat destroyed. In Oaxaca and central Chile, most of the forest fragments are under high pressure from an intensively used matrix, which has led to a progressive transformation towards agriculture and degraded forest respectively (**Fig. 3.10** and **Table 3.2**). Central Chile suffered the largest reduction of forest habitat from 43% to 34% of forest cover and has been severely fragmented (**Fig. 3.12**). In contrast, in Bariloche and Chiapas the native forest fragments remain relatively unmodified and the forest persisted and even defragmented after long periods of grazing by livestock, expansion of crops and fires. Salta was found to be in an intact state in 1977 (94% forest cover) and then changed to a variegated state in 2006 (73%). In this study area, the dryland forest still forms the landscape matrix and is represented by large patches (**Fig. 3.9**). However, forestry operations in these forests impose slight but continuous changes in the spatial patterns of forest cover, reflected by a reduction in the degree of aggregation (**Fig. 3.7**). The identification of the level of modification is an important consideration for management planning, as this can assist in deciding where and when to allocate greater and lesser protection to the landscape (Hobbs, 2002).

Trends in spatial patterns in the dryland forest in Latin America

The analysis of the landscape indices enabled the spatial patterns of dryland forests in Latin America to be assessed. Our results show that the spatial patterns of change of dryland forest were dynamic and did not necessarily represent a unidirectional process of forest loss and fragmentation (**Table 3.2**). This is consistent with what has been described for some other landscapes, emphasizing that there is no single correct way in which to think about spatial patterns in modified landscapes (Lindenmayer and Fischer, 2006). By recognizing the diversity of landscape trends, conservation strategies could be better focused (McIntyre and Hobbs, 1999).

In central Chile, the spatial patterns are related to a unidirectional landscape alteration, with continued alteration assumed to reduce both the size of individual patches (known as shrinkage) and the overall number of patches (known as attrition) (Forman, 1995b) (**Table 3.1**). Veracruz and Salta also showed a unidirectional alteration, but with an opposite trend in the number of patches that increased over time, defined as fragmentation (Forman, 1995a) (**Table 3.2**). On the other hand, Oaxaca showed a bidirectional alteration characterized by a rapid fragmentation in the earliest years followed by a loss of forest patches in the last time interval (**Table 3.2**). This trend was more evident owing to a rapid increase in the number of patches during the first time interval (**Table 3.1**). However, there are many cases where trends in landscape change have been reversed (Metcalfe and Bradford, 2008; Vellend, 2003; Wittenberg *et al.*, 2007). Bariloche showed an increase in patch size and increase in patch proximity over time (**Table 3.3**). In this area, the spatial patterns changed during the study period because of forest regeneration in logged areas. Chiapas showed a more stable pattern of change, increasing slightly the size of forest fragments and reducing the number of patches (**Table 3.1**).

Our results demonstrate that the spatial patterns of dryland forests were highly dynamic over the last four decades. While most of the study areas experienced a reduction of forest habitats others showed an increase or stability in forest cover. Understanding the trends in spatial patterns of dryland forest is important for the conservation of its biodiversity and the provision of diverse ecosystem services. Despite this importance, many forest assessments and international initiatives still focus on the extent of forest loss without concern for its spatial pattern (Kupfer, 2006). This work confirms further the advantages of using landscape metrics to describe pattern change, as has been demonstrated in other parts of the world (Bhattarai *et al.*, 2009; Cayuela *et al.*, 2006; Martínez *et al.*, 2009; Peng *et al.*, 2010; Trani and Giles, 1999; Zeng and Wu, 2005).

Table 3.2 Landscape states and trends in the spatial patterns of dryland forests in six study areas in Latin America over the last four decades.

Study area	Description	Trend in spatial patterns	Landscape state
Salta	Division of large forest patches, increasing number of patches, decrease in forest aggregation	Progressive fragmentation	Intact to variegated
Veracruz			Relictual
Oaxaca	Loss of forest cover; decrease and increase in forest aggregation; substantial changes in matrix composition	Fragmentation followed by deforestation	Fragmented
Central Chile	Loss of forest patches; and forest continuity	Progressive deforestation	Fragmented
Chiapas	No spatial changes in forest cover	Forest persistence	Fragmented
Bariloche	Union of forest patches; increase in forest aggregation	Forest patch coalescence	Fragmented

Mapping forest degradation

Human disturbances not only alter the spatial patterns of forest cover but can also lead to a modification or degradation of the remaining habitat (McIntyre and Hobbs, 1999). Modifications include changes to the structure, biotic composition, or ecosystem functioning of habitat (McIntyre and Hobbs, 1999; Ravi *et al.*, 2010). Degradation of dryland forest habitat is associated with diverse human activities such as livestock grazing, tree harvesting and changed fire regimes (Reynolds *et al.*, 2007).

MODIS data were used to map forest degradation in central Chile between 2002 and 2009 (**Box 3.3**). Owing to the low spatial resolution of MODIS images (250 m), a threshold of $3m^2/m^2$ in Leaf Area Index (LAI) was used to select only dense forests. This enabled other land-cover types such as pasture or shrubland to be excluded in order to monitor the degradation of forest cover. Then, changes in red and infrared band values extracted from MOD 13 Q1 between 2002 and 2009 were analyzed to detect pixels with a degree of degradation. Essentially, it was assumed that dense forest pixels with an increase in reflectance values for the red band and a decline in the infrared band over time correspond to pixels affected by degradation. Pixels without changes in reflectance values for both bands have not been affected by degradation. Degraded forest pixels were selected to determine the degree of degradation. This was conducted applying the NDVI (Normalized Difference Vegetation Index) for which three levels of degradation were defined: NDVI>0.71: low degradation; 0.57<NDVI<0.71: intermediate degradation; 0.57<NDVI: high degradation. These levels were validated in the field. Finally, an assessment of forests degraded during each time interval (2002–2005 and 2005–2009) was obtained by overlapping the corresponding binary maps of degraded forest/non-degraded forest cover in a Geographic Information System (GIS).

A logistic regression modelling approach was used to determine the immediate drivers of degradation. Most of the environmental and socio-economic explanatory variables that are potentially related to forest degradation were identified and mapped. These are: property size, slope, elevation, distance to roads, distance to rivers, distance to towns and distance to agricultural land in the earliest image.

In Chile, the forests degraded between each time interval (2002–2005 and 2005–2009) were identified by overlapping the corresponding binary maps of degraded forest/non-degraded forest cover in a GIS. In Argentina, degradation analysis was conducted for eroded premontane forest for each of the following time intervals: 1977–1987, 1987–1983, 1983–2006 and 1977–2006. The binary response variable, degraded habitat vs. non-degraded habitat, was analyzed using a logistic regression model in the statistical package R (Echeverria *et al.*, 2008). In Salta, Argentina, degraded forest was derived from the Landsat image classification. This corresponded to forest areas with less than 50% of tree cover classified as eroded premontane forest with limiting edaphic factors. This included two types of savannah: (a) with edaphic restrictions and fire controlled, dominated by tusca blanca (*Acacia albicorticata*) and urundel (*Astronium urundeuva*), (b) tusca blanca and pasto cubano savannah, small patches of secondary forest (previously used for agriculture) and low stature riparian forests. The following were also included: eroded premontane forest, secondary forest and riparian forest. Degradation analysis was conducted for eroded premontane forest for each of the following time intervals: 1977–1987, 1987–1983, 1983–2006 and 1977–2006.

A total of 27,831 ha, equivalent to 28% of the total dense dryland forest in 2002, was degraded by 2009 in the Chilean study area (**Fig. 3.11**). Regarding the degree of degradation, the results demonstrated that the proportion of highly degraded forest increased in the second time interval. In 2005, 64% of the degraded forests were categorized as highly degraded, 23% as a moderately degraded and 14% as slightly degraded (**Figs. 3.11** and **3.12**). In 2009, 74% of the degraded forest was highly degraded, 24% moderately degraded and 2% slightly degraded (**Figs. 3.11** and **3.12**). This increase in the area of highly degraded forest can be explained by the fact that local people continued logging the dense forest for firewood and other forest products. These processes have caused a modification of the structure and composition of the remaining forests.

In Salta, 4.8% of the forest cover was degraded in 1977 (**Fig. 3.13**). This value remained almost constant during the following years, reaching 4.6% in 2006 (**Fig. 3.14**). A higher proportion of degraded forest occurred in 1987 with 5.3% of the forest cover. Between 1977 and 1987, forest degradation was mainly associated with livestock grazing, and forest logging for firewood and timber. At the beginning of the 1990s the rapid conversion of degraded forest to soya crops led to a decline in the area of degraded forest. During the last decade, the proportion of degraded forest remained almost constant, which reflects the current presence of degrading activities such as forest logging and browsing by livestock.

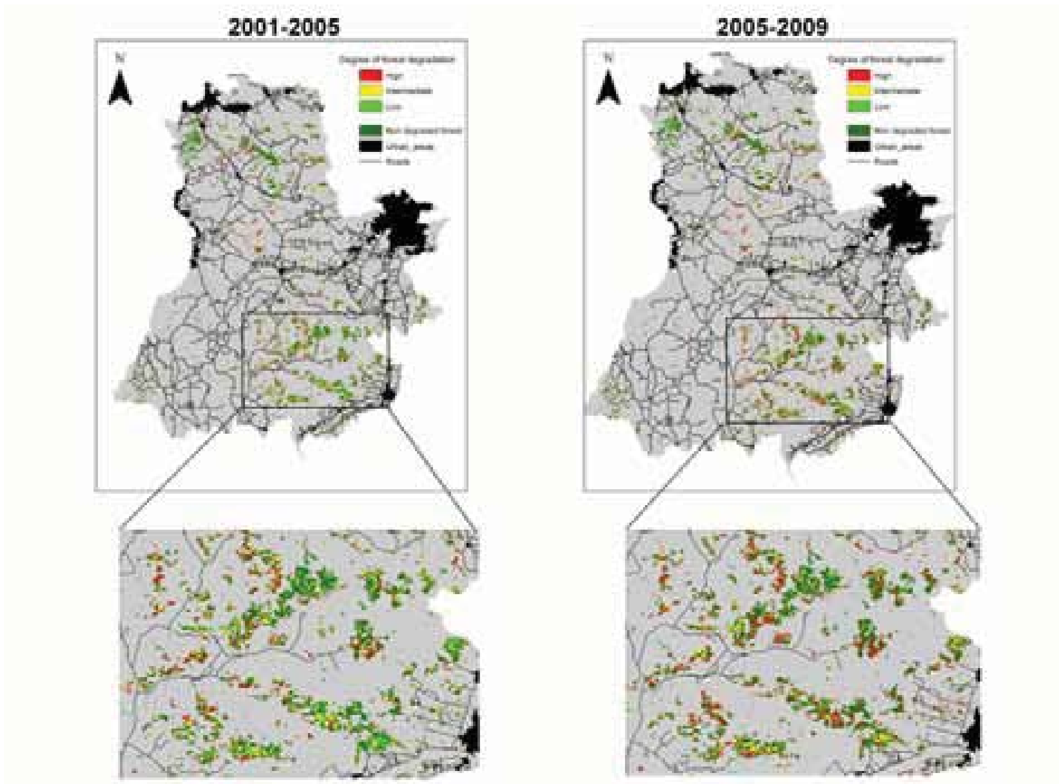

Figure 3.11 Distribution of degraded and non-degraded forest between time intervals in Chile. Degraded forest is shown at three levels of degradation.

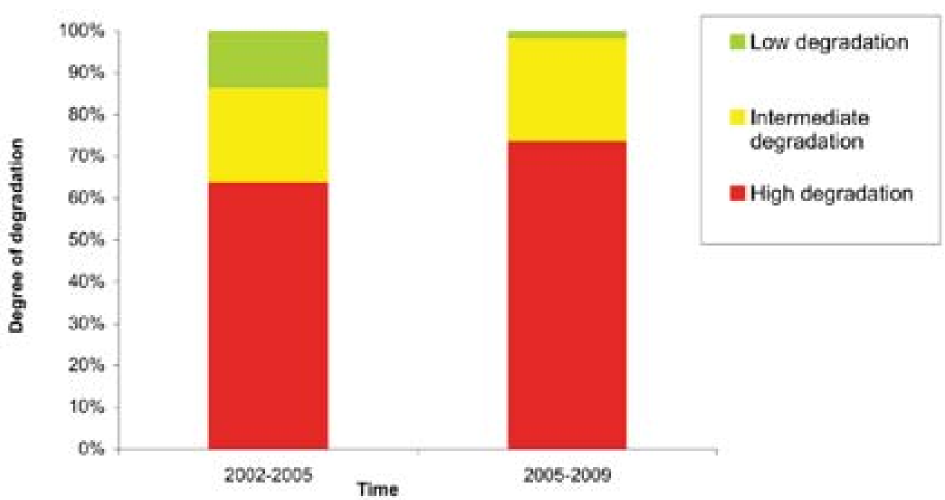

Figure 3.12 Distribution of degraded forest by level of degradation between time intervals in Chile.

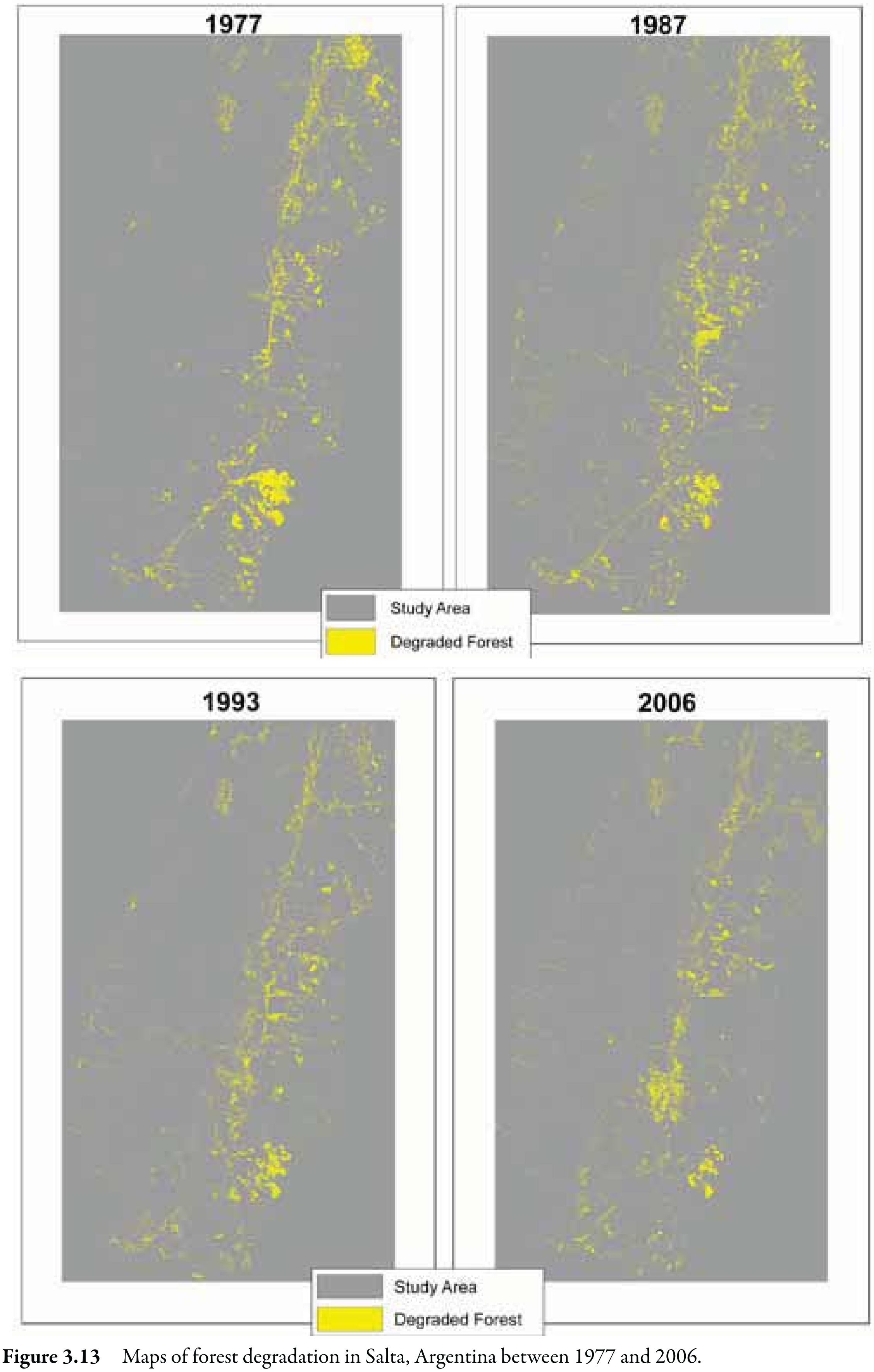

Figure 3.13 Maps of forest degradation in Salta, Argentina between 1977 and 2006.

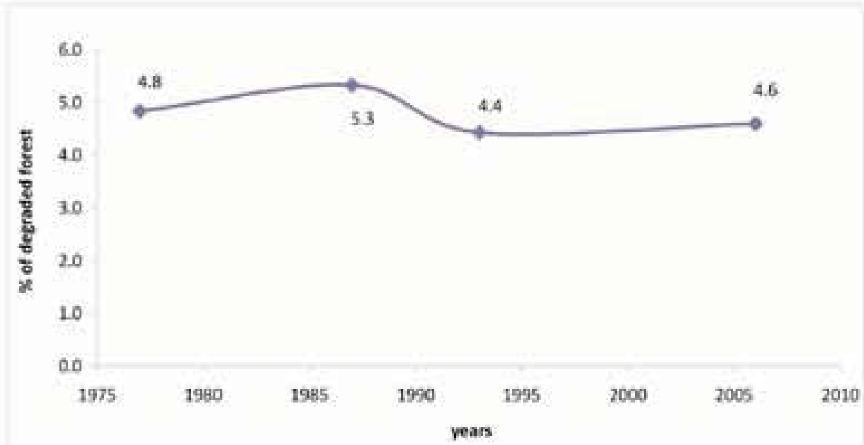

Figure 3.14 Variation in the proportion of degraded forest with respect to the total forest in Salta, Argentina between 1977 and 2006.

Box 3.3 Estimating forest degradation in dryland landscapes in central Chile using MODIS products

D. González, R. Fuentes, C. Echeverría.

Forest degradation may occur owing to natural (forest fire, earthquakes, volcanism, etc.) and anthropogenic perturbations (urban and agriculture surface expansion, forest use, etc.) (Stuart *et al*., 2002; Pickett and White, 1985; Hüttl and Schneider, 1998). Nutritional disturbances can also lead to decreasing stand stability and productivity (Stolpe *et al*., 2008). Although dry forests are being subjected to a range of different disturbances, a major factor responsible for their loss and degradation is the recent expansion of industrial agriculture, resulting from increasing global food demand (Grau *et al*., 2009).

Remote sensing imagery becomes a powerful tool to evaluate the threats to forest ecosystems (Luque, 2000; Armenteras *et al.,* 2003; Echeverría *et al*., 2007). The research described here focused on quantifying the degradation of a dryland forest in an area of 1,250,000 ha using MODIS satellite products. The study area is located in one of the most populated regions of dryland forest in Chile, between 33° and 38° S latitude, located between the Central Valley and Coastal Range (**Fig. 1**).

Using the MOD15A2 product at 1000 m spatial resolution, for three years (2002 (t_0), 2005 (t_1) and 2009 (t_2)), we selected those forest patches whose pixels were equal to or greater than 3 m^2/m^2 of leaf area index (LAI). This threshold enabled dense forest patches that may exhibit degradation to be distinguished. Those patches in which disturbances may have caused a removal of the forest cover were discarded as they represented deforestation instead of degradation.

Further, we applied the Near Infrared (NIR) and Red (R) reflectance responses from MOD13Q1 products at the pixel level to quantify the degradation over time. In a given forest pixel, when the NIR reflectance increases and the R reflectance decreases between two measurements (at the same daily time), the forest is increasing its canopy cover and, therefore, is becoming more dense. On the other hand, when the NIR reflectance goes down and R reflectance goes up, the forest is decreasing its canopy cover, which indicates that the forest patch is being degraded through time (Chuvieco, 1996). These responses in reflectances were modelled in ARC GIS applying a decision tree procedure (**Fig. 2**).

To visualize the levels of forest degradation, we related the Red and NIR behaviour with Normalized Difference Vegetation Index (NDVI) (Eq. 1). This was conducted applying the NDVI where three levels of degradation were defined: NDVI>0.71: low degradation; 0.57<NDVI<0.71: intermediate degradation; 0.57<NDVI: high degradation. All these levels were validated in the field.

Box 3.3 (cont.)

Eq. 1. NDVI=(δNIR-δRed) / (δNIR+δRed), where:

δNIR: Near Infrared reflectance

δRed: Red reflectance

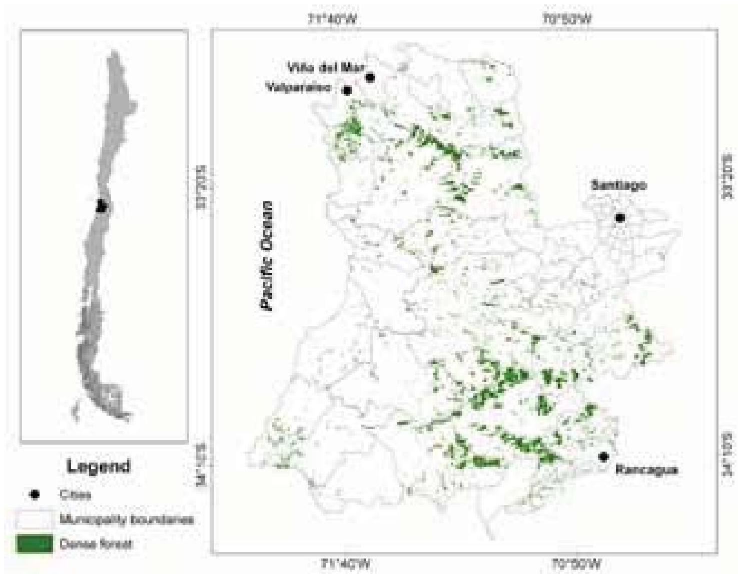

Figure 1 Location of dense dryland forest in the study area in central Chile.

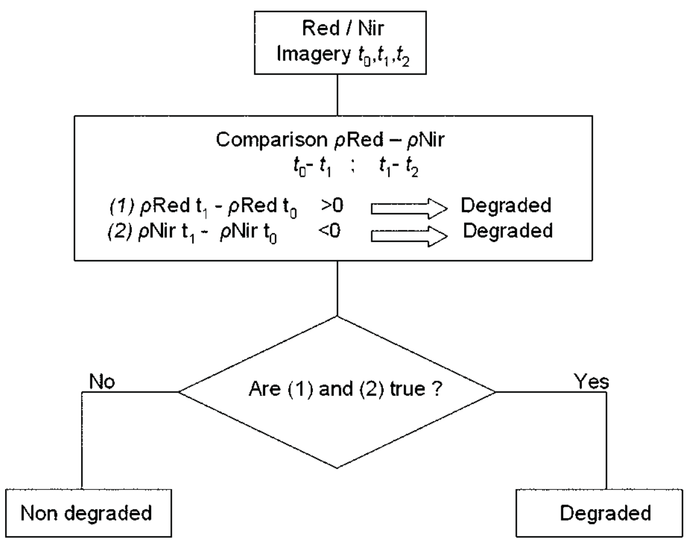

Figure 2 Decision tree for the quantification of forest degradation at the pixel-level.

A logistic regression modelling approach was applied to determine the immediate drivers of degradation. Most of the environmental and socioeconomic explanatory variables that are potentially related to forest degradation were identified and mapped. These include: property size, distance to roads, distance to rivers, distance to towns, distance to agricultural land in the earliest image and slope and elevation. Our results revealed that in 2005, 64% of the degraded forest surface was categorized as highly degraded, 23% as a moderately degraded and 14% as slightly degraded. In 2009, 74% of the degraded forest was highly degraded, 24% moderately degraded and 2% slightly degraded.

Box 3.3 (cont.)

Table 1 Percentage of degraded dense forest at the municipality level. Municipalities of at least 500 ha are listed here.

Municipality	Degradation (%)	Municipality	Degradation (%)	Municipality	Degradation (%)	Municipality	Degradation (%)
Olmué	11.8	Navidad	25.6	Doñihue	28.7	Graneros	33.7
Limache	15.3	Coinco	26.7	San Antonio	29.8	Melipilla	34.6
Quilpue	19.4	Alhué	26.8	Buín	30.1	San Pedro	37.9
Casablanca	20.6	Valparaíso	27.8	Talagante	33.0	María Pinto	38.1
Santo Domingo	23.2	Paine	28.1	Curacaví	33.2	El Monte	42.5
Litueche	24.4	Rancagua	28.2	Las Cabras	33.5	Cartagena	46.9

The municipality with highest forest degradation percentage was Cartagena with 47% (**Table 1**). The municipalities that showed the lowest forest degradation percentages were Olmué (12 %), and Limache (15%) (**Table 1**). The multivariate logistic regression model, used to identify the main drivers of the landscape change process, indicated that the probability of an area being degraded is highly significant and positively related to the distance to streams. In contrast, the distance to urban areas and the distance to agricultural land in 2008 were negatively related to forest degradation. These results showed that the probability of degradation increases in forests located near urban areas, near agricultural land and far away from streams. In these areas, forest logging for fuelwood and browsing by cattle in dense forest are more intense causing a decline in canopy cover and tree density. Human access to dense forest appeared to be one of the main drivers of forest degradation.

The temporal analysis of the forest canopy changes based on NIR and red reflectance behaviour appears to be a suitable procedure to evaluate the degradation of dense dry forests. The main limitation of this procedure relates to the spatial resolution of the MODIS products and the size of dense forest areas.

Causes of forest degradation

In central Chile, the multivariate logistic regression models indicated that the probability of an area being degraded is highly significant and positively correlated with distance to streams (p<0.001; **Table 3.2**). In contrast, distance to urban areas and distance to agricultural land in 2008 were negatively related to forest degradation (p<0.01; **Table 3.2**). These results showed that the probability of degradation increases in forests located near urban areas, agricultural land and far away from streams. In these areas, forest logging for fuelwood and browsing by cattle in dense forest are more intense, causing a decline in canopy cover and tree density. Human access to dense forest appeared to be one of the main drivers of forest degradation.

Similarly, the probability of degradation in Salta was positively related to distance to urban areas in all the time intervals and for the whole study period (p<0.001; **Table 3.2**). Elevation also was positively related to forest degradation in all of the study periods. Distance to streams was marginally significant in the first and the third time interval as well as during the overall study period. This is because most of streams are located in areas that are less accessible for people. The probability of forest degradation was positively explained by distance to villages in the first time interval as well as during the whole study period (1977–2006). Before the 1990s, the distance to secondary roads was significantly related to forest degradation. Since the 1990s this variable has not been significant as most of the degraded forests near secondary roads were converted to agricultural land. During the 1990s, the distance to agricultural land was associated with the presence of degraded forests owing to the use of fire to expand the agricultural frontier.

In both study areas, the results revealed that accessibility to forest areas is one of the main drivers of forest degradation. The probability of a forest area being degraded is higher when a forest is located near urban and agricultural lands, and in lowlands. This trend reflects that the remaining dryland forests in central Chile and in Salta have been undergoing continuous degradation over recent decades (**Box 3.4**). This is consistent with the results obtained by recent assessments (MEA, 2005; Ravi *et al*. 2010), which demonstrate that dryland ecosystems around the world are undergoing rapid land degradation as a result of anthropogenic disturbances. Diverse studies emphasize that modifications of dryland forest habitats may lead to changes in ecosystem processes (Jafari *et al*., 2008; Smet and Ward, 2006; Stolpe *et al*., 2008), which may affect the productivity of the landscape, with important environmental and socioeconomic implications.

Box 3.4 Human-caused forest fires in Mediterranean ecosystems of Chile: modelling landscape spatial patterns of forest fire occurrence

A. Altamirano, C. Salas, V. Yaitul, A. Miranda, C. Smith-Ramírez

Fire disturbance is recognized as an important problem because it can devastate natural resources and human property, and threaten human lives. Forest fires result in enormous economic losses because they affect environmental, recreational, and amenity values as well as consume timber, degrade real estate, and generate high costs of suppression. Modelling forest fire occurrence (i.e., where and when a forest fire starts) has recently been conducted in the northern hemisphere (Calef *et al.*, 2008; Lozano *et al.*, 2007; Ryu *et al.*, 2007; Vega-Garcia and Chuvieco, 2006), however, efforts on the subject are lacking for the southern hemisphere, in particular for Chilean ecosystems. Some studies in Chile have focused on post-fire effects on vegetation dynamics (Navarro *et al.*, 2008; Litton and Santelices, 2003), but studies on predicting forest fires occurrence are lacking. In Chile it has been reported that fire can encourage exotic plant invasions (see Chapter 8) and cause significant losses of local biodiversity. Forest fire occurrence has increased in recent years in Chile, with a mean frequency of about five thousand forest fires per year. These fires have affected a mean area of about 500 km^2 per year (Navarro *et al*., 2008; CONAF, 2009), human activity being the main cause of fire ignition (CONAF, 2009). The extensive fires produced by human activity in central Chile need to be addressed in order for forest restoration approaches to be effective.

It is important to understand the impact of processes such as fire ignition and spread on landscape patterns in order for land management practices to be effective (Foster *et al*., 1997). At the landscape scale (i.e. extents >100,000 ha), the probability of a large fire is associated with multiple factors including: forest type, physiographic characteristics, climate, and human activities. In this study we developed models to investigate the relationship between forest fire occurrence and landscape heterogeneity spatial patterns in Mediterranean ecosystems of Chile. The study area extends over 892 km^2 and is located in eastern central Chile covering parts of the Valparaíso and Metropolitan administrative regions (**Fig. 1**). We selected a landscape with temporal stability in composition operating at the landscape scale. We used georeferenced forest fire data from a 5-year period of fire occurrence from 2004 to 2008. A distance of 25 x 25 pixels (750 m) was used to compute the co-occurrence matrices, since small windows result in very sparse matrices. Our data on landscape spatial patterns were obtained at multiple spatial scales, including climatic, topographic, human-related, and land-cover variables from satellite imagery. We fit a logistic model in order to predict forest fire occurrence as a function of our potential predictor variables. The relationship modeled was that between the binary response variable (one = burned, zero = not burned) and the predictor variables. In order analyze forest fire occurrence we produced categorized maps of the predicted forest fire occurrence probability into four levels: very high ($0.75 \leq p \leq 1$), high ($0.5 \leq p \leq 0.75$), low ($0.25 \leq p \leq 0.5$) and low ($0 \leq p \leq 0.25$).

Our best model suggests that the probability of forest fire occurrence is related to both high temperature and precipitation, and lower distance to cities. Our predictions suggest that 46% (410 km^2) of the study area has high probability of forest fire occurrence, being concentrated in the eastern locations of the study area (**Fig. 1**). Our model correctly classified about 73% of our validation dataset. The information from this study may be useful for hazard reduction, indicating risk of forest fire occurrence (Ryu *et al*., 2007; Vega-Garcia and Chuvieco, 2006). The study area is one of the most populated regions of Chile. Therefore, our findings can be used to inform decision making regarding land and urban planning. If climate determines patterns of forest fire occurrence, then when the climatic variables change, forest fire occurrence may also change. This might have important consequences for long-term land and urban planning, since prioritization of areas with high probability of forest fire occurrence today might not be effective in the face of climate change. Exploring a new statistical model approach would allow to improve the predictive capability of the models. Therefore, part of our future research will focus on this subject.

Box 3.4 (cont.)

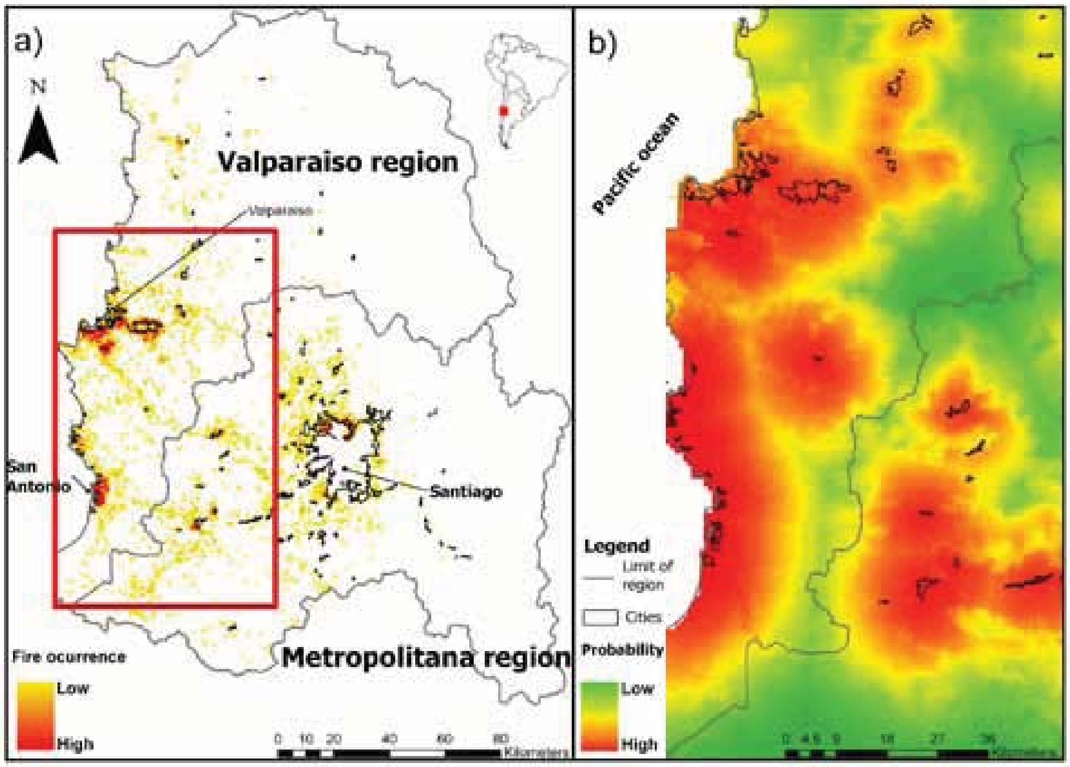

Figure 1 (a) Map of study area and records of forest fires between 2004 and 2008, and (b) map of forest fire occurrence probability based on the predictive model.

Conclusions

Our results indicate that dryland forests exhibit a progressive fragmentation and degradation in most of the Latin American landscapes studied during the research. In central Chile and in Salta, dryland forests have been simultaneously affected by forest loss (Chapter 2), fragmentation and degradation. In Veracruz and Oaxaca, the landscape has experienced a continuous fragmentation and loss of forest habitats. On the other hand, Chiapas and Bariloche show different trends, towards forest persistence and coalescence respectively. Results presented here clearly show that dryland forest is under considerable human pressure from economic development and imply policy challenges for the countries involved. Owing to the importance of dryland forest for providing different ecosystem services for human well-being, diverse actions should be undertaken to minimize or reverse the human impacts of fragmentation and degradation on dryland forests. Ecological restoration actions have the potential to address both the fragmentation and degradation of forest that has been documented here in multiple study areas. Such interventions should be planned and implemented at the landscape scale, to ensure they are effective in increasing connectivity among forest patches. Recent advances emphasize the development of integrative approaches to counter land degradation, poverty, safeguard biodiversity and protect the culture of the 2.5 billion people who live in dryland systems (Reynolds *et al.*, 2007). Forest landscape restoration actions should constitute an element of such approaches. Urgent and comprehensive reframing of rural development strategies in Latin America should be undertaken to achieve this goal.

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4 FRAGMENTATION AND ALTITUDINAL EFFECTS ON TREE DIVERSITY IN SEASONALLY DRY FORESTS OF MEXICO AND CHILE

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Introduction

Changes in the number of plant or animal species in relation to area of habitat patch (Harris, 1984), latitude and, its mirror image elevation (Rahbek, 1997), provide the best-known relationships describing natural patterns of species richness for a number of diverse taxa (Connor and McCoy, 1979; Rohde, 1992). In areas historically subjected to strong and persistent anthropogenic disturbance, such as tropical dry forests (TDF) and Mediterranean climate woodlands, variables associated with human impact, such as fire, cattle grazing and logging, can greatly influence species diversity patterns to TDF (e.g. Bullock *et al.*, 1995; Gentry, 1995; Trejo and Dirzo, 2002; Segura *et al.*, 2003; Gordon *et al.*, 2004; White and Hood, 2004; Balvanera and Aguirre, 2006; Williams-Linera and Lorea, 2009) and Mediterranean climate woodland (e.g. Bond, 1983; Armesto and Martínez, 1978). Other physical variables expected to be associated with patterns of tree species richness in seasonally dry forests are the length of the dry season and the amount and timing of precipitation (Richerson and Lum, 1980).

Some authors have shown that tree species richness is not always significantly correlated with the quantity and seasonality of precipitation in TDF (Lott *et al.*, 1987; Gentry, 1995; Gillespie *et al.*, 2000; Trejo and Dirzo, 2002). Instead, patterns of plant species diversity have been found to be associated with variation in potential evapo-transpiration (Trejo and Dirzo, 2002), and with differences in soil moisture availability in relation to elevation, insolation, slope, and soil water-holding capacity (Balvanera *et al.*, 2002; Segura *et al.*, 2003; Balvanera and Aguirre, 2006). Based on research undertaken in Mexico, Balvanera and Aguirre (2006) reported that different tree species occupied different parts of the soil moisture gradient, and that many species were excluded from the driest sites, where productivity was lowest. In another Mexican study, Segura *et al.* (2003) reported that tree species richness declined as soil water availability decreased along a 1 km-long watershed, showing that drier conditions tend to support lower plant species richness. In these forests, live stem densities increased substantially with water availability, while the proportion of dead stems increased towards the drier end of the gradient.

Many of the remnants of dry forests have been subjected to anthropogenic fragmentation, and hence abiotic changes that result from reductions in forest patch area are likely to produce declines in local diversity and density of native trees and other species (e.g. Bennett, 2003; Cadenasso and Pickett, 2001; Echeverría *et al.*, 2006; Drinnan, 2005; Hersperger and Forman, 2003; Hobbs, 2001; Holt *et al.*, 1995; Honnay *et al.*, 1999; Laurance *et al.*, 1998a, 1998b, 2001; Matlack, 1994; Quinn and Harrison, 1988; Simonetti *et al.*, 2001; Soulé *et al.*, 1992; Tabarelli *et al.*, 1999; Willson *et al.*, 2001). However, the typical effects of patch area on species richness have not always been found, for example in the case of South African Mediterranean woodlands (Kemper *et al.*, 1999). This is due to the fact that tree species richness is not only influenced by the area of remnant forest patches, but also by other patch-related variables, such as edge effects, distance to roads or history of human impact. Variables such as the proportion of core area, perimeter, shape, and connectivity of fragments can also have important consequences for biodiversity (Forman and Godron, 1986; Drinnan, 2005). Recent studies indicate that the spatial configuration of remnant patches may also influence species richness of herbaceous plants (Petit *et al.*, 2004). No studies have explicitly examined the influence of remnant patch attributes on tree diversity patterns in dry tropical and Mediterranean forests (but see Kemper *et al.*, 1999).

This study describes and compares the combined effects of elevation and forest patch attributes on tree species richness and composition in four representative, seasonally dry forests in the Americas (see also **Boxes 4.1 – 4.8** for associated studies). All landscapes studied have been greatly transformed by human activities. The study areas were located in the dryland tropical forests of southern Mexico (three areas) and the Mediterranean woodlands of central Chile. We present statistical models of environmental and patch variables that account for the tree diversity patterns observed. The aim of this research was to examine the factors influencing patterns of species richness in fragmented dryland landscapes, with the aim of informing approaches to forest landscape restoration. To be effective in restoring biodiversity, such approaches will need to be based on a firm understanding of the processes influencing species richness patterns.

Study sites, sampling and statistics

Veracruz, Mexico

The study area was central Veracruz in the adjacent municipalities of Comapa and Paso de Ovejas (19° 17' N and 96° 26' W, between 97 and 420 m elevation), covering an area of 300 km². Mean minimum and maximum temperatures are 19.8°C and 30.7°C, respectively. Mean annual precipitation is 966 mm (range: 502–1466 mm), which is unevenly distributed during the year. The dry season extends from October to May (station at Loma Fina; 7 to 28 km from the study sites). Land use in this region is dominated by small-scale cattle ranching by private landowners, but communal tenants (*ejidatarios*) practice more diverse land uses, mainly maize farming (Gallardo-López *et al.*, 2002). Some dominant tree species are *Bursera cinerea*, *Calyptranthes schiediana*, *Comocladia engleriana*, *Ipomoea wolcottiana*, *Leucaena lanceolata*, *Luehea candida*, *Savia sessiliflora*, *Spondias purpurea*, *Tabebuia chrysantha* and *Thouinidium decandrum*. Ten forest fragments were selected to characterize the forest of Paso de Ovejas (Williams-Linera and Lorea, 2009). The study sites were located 0.5 to 22 km away from each other (mean = 10.7 km); the fragments are believed to be remnants of a once continuous forest cover.

Oaxaca, Mexico

In Oaxaca, study sites were located in the municipalities of Santiago Apoala (17°33'N–97°5'W), Santiago Huauclilla (17°25'N, 97°1'W) and Santiago Tilantongo (17°3' N, 97°17'W) (see also **Box 4.4**). The climate is sub-humid to moderately semi-dry, annual rainfall ranges from 600–700 mm (Santiago Apoala) to 800–1000 mm (Huauclilla; Tilantongo). In each area, we selected four forest fragments with a relatively homogeneous vegetation cover. Fragment selection was aimed at obtaining ecologically contrasting areas. The fragments were selected by using 2005 SPOT satellite images, using ArcView® to detect masses of homogeneous vegetation, separated by at least five pixels (150 m distance) from similar vegetation patches (**Fig. 4.1**). Fragment metrics including area, perimeter, and perimeter/area ratios were quantified using FRAGSTAT® (McGarigal and Marks, 1994). A total of 216 circular plots (5.7 m radius, 102.1 m^2) were sampled. The plots were randomly distributed and separated from each other by a minimum distance of 70 m. Each tree (>2 m in height and >2.5 cm stem diameter) was positioned by means of polar coordinates. When trees could not be identified taxonomically, they were assigned to morphospecies categories.

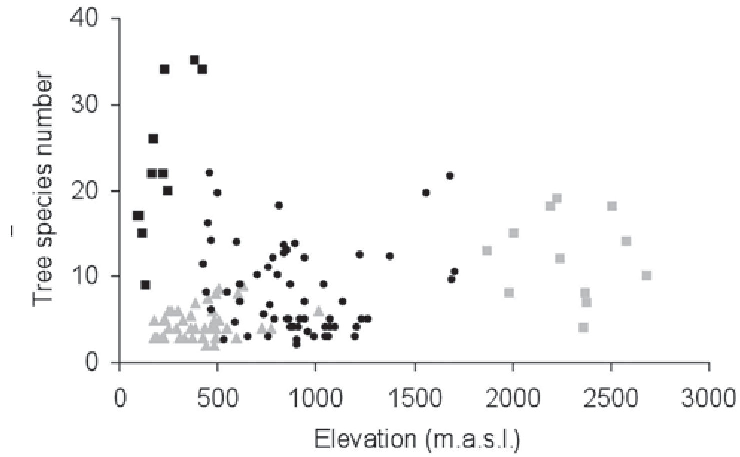

Figure 4.1 Seedling abundance in relation to changes in elevation of remnant fragments of seasonally dry forests in Mexico and Chile. Veracruz (dark squares), Oaxaca (dark circles) and Chile (gray triangles).

Chiapas, Mexico

The study area was located in the Central Depression of Chiapas between 17° 59'–14° 32' N, and 90° 22'–94° 14' W. The region is completely surrounded by warm, moist mountain areas in transition zones with colder and drier woodlands that provide a complete isolation from other regions with TDF. The vegetation was originally dominated by tropical deciduous vegitation, but much of this has been replaced by secondary vegetation owing to grazing, fire, construction of dams and cutting. Only in certain protected areas such as canyons and upper slope areas are strongholds of vegetation present that exhibit relatively little evidence of human influence. At upper elevations, the climate is driest with a mean annual temperature of 22.8°–25.8 °C and an average rainfall varying between 660 and 1051 mm annually. At the middle and upper elevations (>800 m a.s.l.) the climate is semi-warm and dry, with the rainfall varying between 1110 and 1267 mm and a mean annual temperature below 23°C. Site selection was preferential (Matteucci and Colma, 1982), and was performed using high resolution satellite images in Google Earth 4.3 supported by field surveys, to identify the present vegetation physiognomy in the study area. All individual trees >10 cm dbh were counted in 1000 m^2 circular plots, and individuals between 5–10 cm dbh (diameter at breast height) were counted in an inner 100 m^2 circular plot. The number of plots surveyed in each forest fragment was determined by the structural complexity (the degree of homogeneity, taking into account the dominant species) and spatial extent.

Central Chile

The study was conducted on the coastal range of central Chile (between 32º–34ºS), between 300 and 1200 m elevation, covering an area of about 7000 km^2. Mean minimum and maximum temperatures are 4.5°C and 33.4°C, respectively (di Castri and Hajek, 1976). The climate is characterized by a dry season of 5 to 7 months during the austral summer (October to April), with a mean annual air temperature of 12.7°C (di Castri and Hajek, 1976). Mean annual precipitation ranges from 350 mm to 800 mm, increasing from north to south (di Castri and Hajek, 1976), and rain is concentrated in relatively few events that occur during the austral winter (May to August). Remnant patches of sclerophyllous woodlands in this region are often surrounded by exotic conifer, or eucalypt plantations, highly degraded shrublands, or anthropogenic grasslands. Dominant tree species in remnant woodlands include shade-tolerant species such as *Cryptocarya alba*, *Peumus boldus*, *Dasyphyllum excelsum*, *Beilschmiedia miersii*, and markedly shade-intolerant tree species such as *Quillaja saponaria*, *Schinus latifolius*, and *Acacia caven*. The Mediterranean-climate region of central Chile is characterized by strong abiotic and vegetational heterogeneity, especially between contrasting slopes of north vs. south aspect (Armesto and Martínez, 1978).

Selection and digitization of woodland patches was carried out from Google Earth images. Fragments were classified into four size classes (0.5–10 ha, 10–100 ha, 100–1,000 ha, and >1,000 ha), representative of the full range of remnant woodlands in the landscape. The number of patches sampled varied for each size class: 22, 9, 7 and 3 patches, for each of the above size classes respectively. Fragments above 1200 m elevation were not sampled, to exclude non-sclerophyllous forests. We controlled for slope aspect effects by sampling on slopes with south, southwest and/or southeast exposure. Plots were predominantly located on slopes between 15 and 35 degrees. In each fragment we recorded the number of tree species, densities of stems >5 cm, diameter at breast height (dbh) and number of seedlings (0.1 cm–2 m tall) present within one (size classes 1–3) or two (size classes 4–5) 10 x 10 m plots. This number of plots provides a reliable estimate of tree species richness in the entire patch, according to species/area curves (Becerra *et al.*, unpublished data).

Statistical analysis

Dependent variables were tree and seedling species richness and abundance. In Chiapas, seedling abundance was not measured. Independent variables were elevation, fragment area, perimeter and perimeter/area ratio. Patch areas and perimeters were significatively correlated in Veracruz (r2 = 0.75, p = 0.08), Oaxaca (r2 = 0.94, p <0.001) and Chile (r2 = 0.96, p <0.001), and hence only area was analyzed as a patch variable and elevation as an environmental variable. Statistical analyses were conducted using area and log area, but since no differences were found between these approaches, only linear relationships are reported. To assess the effect of patch area and elevation on tree and seedling species richness and abundances, a GLIM model selection was conducted using R project software (version 2.7.1, 2008) (R Development Core Team, 2005), with log transformed data to correct for departure from normality, and including the effect of location and country as co-variates. We selected models with and without interactions among factors, based on the lowest AIC (Δ AIC >2). Otherwise models were considered statistically identical, in which case we selected the model with the smallest number of parameters, assuming a parsimonious criterion (Burnham and Anderson, 2002).

Cleared dry forest in Veracruz, Mexico. Photo: C. Alvarez

***Acacia spp*., Chile. Photo: C. Echeverria**

Results and discussion

Patterns of tree species richness

In Veracruz, Mexico, a total of 175 species of adult trees and 60 species of tree seedlings were recorded in a total of 11 remnant fragments of tropical dry forest that were sampled (**Box 4.8**). In Oaxaca, southern Mexico, a total of 52 species of adult trees and 140 species of tree seedlings were recorded in 12 tropical dryland forest fragments. For TDF in Chiapas, a total of 263 tree species were recorded in all forest fragments surveyed (see also **Box 4.2**). In contrast, for Mediterranean sclerophyllous forests in central Chile a much lower total of 14 species of adult trees and 15 species of tree seedlings were recorded in a total of 41 remnant forest fragments.

Pearson correlations were used to analyze the relationships between elevation above sea level and adult tree species richness in all regions, which was positive and significant only for Veracruz (**Table. 4.1**; see also **Box 4.1**). Both in Veracruz and Oaxaca elevation was also related to the overall abundance of adult trees. Elevation was significantly and positively related to tree seedling abundance in remnant forest patches in Oaxaca only (**Fig. 4.1**). Forest fragment area was related to species richness of adult trees only in Oaxaca, and to tree seedling abundance only in Chile. In Chiapas, there was no relationship between forest patch area and adult tree species richness.

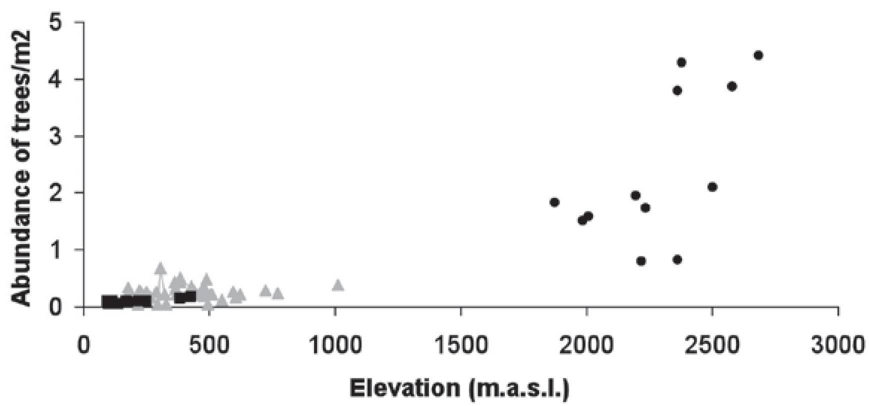

Figure 4.2 Changes in tree species richness (adult) with elevation on remnant fragments of seasonally dry forests in Mexico and Chile. Veracruz (dark squares), Chiapas (solid circles), Chile (gray triangles), Oaxaca (gray squares).

Table 4.1 Pearson correlation coefficients (R) in tropical and Mediterranean dry forests between independent variables (elevation and patch area) and tree species richness (S), density (D) and seedling densities (d). Correlations with p <0.05 are indicated with *, p <0.01 **, ns = non-significant.

	Veracruz	*Oaxaca*	*Chiapas*	*Chile*
Elevation and S	0.83*,	−0.10 ns,	−0.12 ns,	0.26 ns,
Area and S	−0.2 ns	0.77*	−0.16 ns	0.16 ns
Elevation and D	0.85*	0.62*	—	—
Area and D	0.30 ns	−0.38 ns	—	0.04 ns
Elevation and d	−0.22 ns	0.67*	—	0.20 ns
Area and d	−0.44 ns	0.44 ns	—	0.36*

Box 4.1 Altitudinal variation in vegetation structure and diversity of tree species in the tropical dry forest region of central Veracruz

M. Toledo Garibaldi, G. Williams-Linera

In central Veracruz, from the coast of the Gulf of Mexico to the summit of Cofre de Perote (4282 m a.s.l.) located at the end of the Neovolcanic Transversal belt of Mexico, vegetation types range from coastal vegetation to dry forest, cloud forest, oak forest, coniferous forest, high elevation grassland, and alpine vegetation. This continuous change in plant communities along altitudinal gradients has led to several hypotheses – including many related to topography, soil, and climate – to explain the changes in biodiversity and structural patterns of some forests. In this study, we investigated whether the tropical dry forest displayed patterns of variation along the altitudinal gradient. The objective was to determine changes in the vegetation structure and tree species composition of dry forest as they relate to climatic variation.

The study area is located in the tropical dry forest (TDF) region of central Veracruz, Mexico, in the municipalities of Paso de Ovejas and Emiliano Zapata. In the study area, eight sites were selected between 100 and 1000 m altitude in the dry and sub-humid climatic zone. The criteria were that forest fragments be relatively undisturbed and located along the 19th century Royal Road from Veracruz to Mexico City.

In each forest fragment, trees >5 cm dbh (diameter at 1.3 m) were measured and identified to species level on ten 10 x 10 m plots. Precipitation and temperature were obtained from the nearest meteorological stations. Alpha diversity of richness (number of species) and Shannon's diversity index were calculated for each site. Beta diversity was analyzed using the complementarity of fragment richness, which is the proportion of all species at two sites that occur in only one or the other; it varies from zero (when the lists are identical) to one (when the lists are completely different). Basal area, density, and vegetation height were analyzed using ANOVA with post hoc Tukey's significant difference test. The correlation between pairs of variables was determined using Pearson correlation coefficients.

Precipitation and temperature were positively ($r = 0.67$, $p = 0.07$) and negatively related to elevation ($r = -0.97$, $p < 0.0001$), and both are therefore important environmental factors influencing plant species distribution in the altitudinal gradient. The vegetation structure of TDF sites was heterogeneous (**Table 1**). Changes in tree density were significant among sites ($F = 3.40$, $p = 0.004$), and the site at the lowest altitude showed the lowest density (**Table 1**). Vegetation height also changed among sites ($F = 5.75$, $p < 0.0001$), with a trend for lower tree heights at sites with lower elevations (**Table 1**). Tree density was positively related to elevation ($r = 0.63$, $p = 0.09$) whereas the height of the trees was lower at higher altitudes ($r = -0.63$, $p = 0.09$) and tree height and temperature were also positively correlated ($r = 0.69$, $p = 0.06$). In contrast, changes in basal area were significant among sites ($F = 4.18$, $p = 0.0007$) but were not correlated to altitude or to precipitation or temperature (**Table 1**).

A total of 136 overstory tree species were found at the study sites. According to importance values indices (IVI), some of the dominant tree species were *Bernardia mexicana, Bursera simaruba, Caesalpinia cacalaco, Ceiba aesculifolia, Comocladia engleriana, Croton reflexifolius, Ipomoea wolcottiana, Lysiloma acapulcense, Leucaena lanceolata,* and *Piscidia piscipula*. Richness varied between 17 and 34 tree species, and Shannon's diversity index varied between 2.26 and 3.13 (**Table 1**) without a clear altitudinal trend, although they were significantly correlated ($r = 0.86$, $p = 0.006$).

Beta diversity as a measure of complementarity indicated a high species turnover among sites. Nearby sites shared only a few species (76 to 94%). *Bursera simaruba* was the only species distributed along the entire altitudinal gradient; some species appeared to be restricted to one site, but no clear altitudinal trend was detected. The lowest similarity in tree species was between site 8 and the other sites (96 to 100%). The site at the highest altitude was different since it was not TDF: species such as *Quercus sapotifolia* and *Clethra macrophylla* became dominant. TDF and hot sub-humid climate reached its limit at this elevation of 986 m, where the ecotone between TDF and the upper vegetation type – tropical montane cloud forest – occurs.

Box 4.1 (cont.)

TDF had a very heterogeneous vegetation structure; a consistent pattern throughout the altitudinal gradient was not found, although at a lower elevation, the temperature was higher, tree density was lower, and trees were taller. Species composition, on the other hand, did not follow a trend; few species were present at all sites, and others were found only at a few. We concluded that the altitudinal range occupied by TDF is wide in central Veracruz. Within this altitudinal range, changes in structure and tree species dominance from one site to another may be related to factors such as topography, slope, soil type, or anthropogenic disturbance (see **Box 4.9**).

Table 1 Characteristics of the eight study sites located between 100 and 1000 m a.s.l. in central Veracruz, Mexico. Variables are altitude (m a.s.l.), total annual precipitation (mm), mean annual temperature (T°C), basal area (m^2/ha), density (trees/ha), mean vegetation height (m), richness or total number of species, and Shannon's diversity index for trees >5 cm dbh. Values in the same row accompanied by different superscript differ significantly at alpha <0.01.

	1	*2*	*3*	*4*	*5*	*6*	*7*	*8*
	Puente Nacional	*Don Tirzo*	*La Virgen*	*Plan del Río*	*Dos Caminos*	*Cerro Gordo*	*Corral Falso*	*Lencero*
Altitude	140	204	227	335	376	501	780	986
Precipitation	1186	890	890	912	1045	892	1112	1421
T°C	27.2	24.9	24.9	25.1	24.4	23	21.1	19.6
Basal area	29.34^{ab}	20.87^{b}	21.94^{b}	17.54^{bc}	30.32^{ab}	39.19^{a}	22.62^{b}	28.05^{ab}
Density	844^{b}	1100^{a}	1020^{a}	1487^{a}	1320^{a}	1378^{a}	1244^{a}	1470^{a}
Height	10.0^{a}	8.7^{a}	8.5^{ab}	8.0^{b}	9.5^{a}	8.7^{a}	6.78^{bc}	8.1^{b}
Richness	17	34	22	26	33	31	21	29
Shannon Index	2.26	3.13	2.49	2.68	3.01	2.79	2.34	2.37

Box 4.2 Diversity of woody vegetation in the Central Depression of Chiapas, Mexico

N.E. Taylor-Aquino, N. Ramírez-Marcial, R.A. Vaca

The analysis of the patterns, causes and maintenance of tropical biodiversity are issues that have generated considerable attention for many years among biologists and ecologists (Bullock *et al*., 1995). Understanding the spatial variation of plant species diversity is particularly relevant in the tropics because of their high diversity and threatened status (Lawton *et al*., 1998). Generating more biological and ecological information is necessary for developing such an understanding. This information is still scarce for most tropical regions, but is needed to support land-use planning, monitoring, and the development of restoration plans in degraded areas (Lindenmayer and Franklin, 2002; Huston, 2004).

Chiapas is the second most floristically diverse state of Mexico, with a high number of endemic plants (Miranda *et al*., 1963; Rzedowski, 2006; Breedlove, 1981). Its latitudinal range, topography and geological history determine a high spatial environmental heterogeneity and create a large variety of ecological conditions (Breedlove, 1981). Climate is considered as the major determinant

Box 4.2 (cont.)

of vegetation distribution (Woodward, 1987), and the most influential factor on the climate of this region is the topography. Local micro-climates are very fine-grained.

The Central Valley of Chiapas, also known as Central Depression of Chiapas, is located in the central portion of the state. The dry forests of the Central Valley are completely surrounded by moist forested mountain areas, providing relative isolation from other areas of dry vegetation. In this context, dry forests extend over altitudinal gradients, ranging from lowland deciduous and sub-deciduous tropical forests up to oak and pine-oak forests (Breedlove, 1981). Extensive cultivation and grazing has led to large tracts of thorn woodland and savannah (Breedlove, 1981; Challenger, 1998). Following these altitudinal gradients and according to the WorldClim database, the mean annual temperature ranges from 22 to 25.2 °C, and annual precipitation ranges from 750 to 1500 mm. In this work we aimed to identify and describe tree-species associations of dry forest occurring along altitudinal gradients in the Central Valley of Chiapas.

We evaluated the floristic composition and structure of woody vegetation by 131 circular plots (13.1 ha total) in different localities of the Central Depression of Chiapas. The sample was stratified into two diameter-size categories: (1) small trees (individuals 5–10 cm dbh, in plots of 0.01 ha), and (2) large trees (individuals with dbh >10 cm in plots of 0.1 ha). Although this geographical region has a long history of land-use activities, there are still some remnants of woody vegetation in varying stages of successional development. The plots were located in these remnants, over a wide range of environmental variation following altitudinal gradients (440–1740 m).

We recorded a total of 263 tree species distributed in 161 genera and 66 families. Through a Cluster Analysis we identified a total of four tree species associations (**Fig. 1**), based on species dominance: (1) *Matayba oppositifolia-Ternstroemia tepezapote-Tapirira mexicana*, related to tropical sub-deciduous forest (other dominant species: *Nectandra salicifolia* and *Bursera simaruba*); (2) *Bursera simaruba-Cochlospermum vitifolium*, related to tropical deciduous forest (other dominant species: *Heliocarpus reticulates*, *Leucaena shannonii* and *Bursera excelsa*); (3) *Quercus segoviensis-Quercus crispipilis*, related to oak forest (other dominant species: *Ternstroemia tepezapote*, *Rhus schiedeana* and *Quercus polymorpha*); and (4) *Quercus peduncuaris-Quercus acutifolia-Pinus oocarpa*, related to pine-oak forest (other dominant species: *Byrsonima crassifolia*, *Quercus castanea* and *Quercus conspersa*). These groups were arranged along altitudinal and geographical gradients. *Bursera simaruba, Cochlospermum vitifolium, Leucaena shannonii, Heliocarpus reticulatus, Calycophyllum candidissimum,* and *Bursera bipinnata* were the most abundant species in the lowest elevations (under 900 m a.s.l.). In the highest elevations (above 800 m a.s.l.) the most abundant species were *Quercus* (mostly *Quercus peduncularis*) and *Pinus oocarpa*.

We identified four more groups for which distribution was better explained by human influence than climate variables; 85% of their species had less than 10 cm DBH and were pioneer trees: *Ficus pertusa, Mimosa tenuiflora, Heliocarpus reticulatus, Acacia cornigera, Diphysa robinioides, Guazuma ulmifolia, Bursera simaruba, Ficus cotinifolia, Luehea candida, Genipa americana, Casearia corymbosa, Alibertia edulis and Stemmadenia obovata*. Most of these species are found within open pasturelands or may be used as a protein supply for cattle and live fences.

Although regional and beta diversity are relatively high, local diversity is relatively low and is represented by initial and secondary pioneer species, which pre-supposes a long history of changes associated with extreme weather events and interaction with anthropogenic factors.

Box 4.2 (cont.)

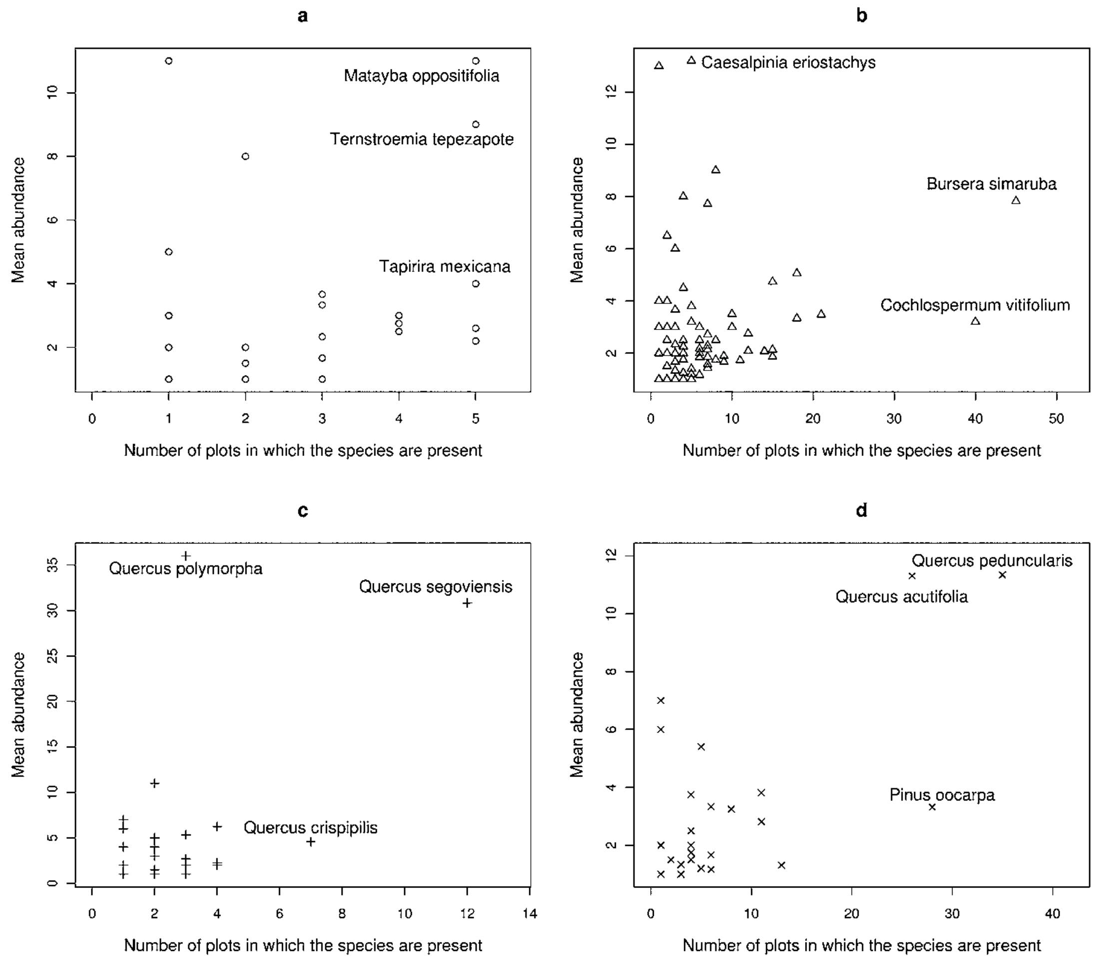

Figure 1 Dry forest associations in the Central Valley of Chiapas: (a) *Matayba oppositifolia-Ternstroemia tepezapote-Tapirira mexicana*, (b) *Bursera simaruba-Cochlospermum vitifolium*, (c) *Quercus segoviensis-Quercus crispipilis*, and (d) *Quercus peduncularis-Quercus acutifolia-Pinus oocarpa*. The dominant species are shown for each association. The y axis represents the mean abundance of species obtained by dividing the total abundance of each species by the number of plots where they were present. These graphs illustrate different aspects of floristic composition: (1) most of the species are not abundant and are distributed in only a few plots, and (2) some of the most abundant species show a reduced distribution, for example *Caesalpinia eriostachys* in plot b, and *Quercus polymorpha* in plot c.

Deforestation of dry forest in central Veracruz, Mexico. Photo: C. Alvarez

Dry forest in Chiapas, Mexico. Photo: N. Ramírez-Marcial

Box 4.3 Tree species diversity and forest structure in subtropical dry forest of northwestern Argentina

C. Blundo, L. R. Malizia

Subtropical seasonally dry forests (SSDF) in northwestern Argentina (22°–24°S and 63.5°–65°W) include Chaco forest (CF) from 300–400 m a.s.l. and premontane forest (PF) from 400–900 m a.s.l. PF has usually been called 'transitional forest' (Hueck, 1972) between the neighbouring dry CF and the more humid Yungas forest (i.e. *Selvas de Montaña*, Cabrera 1976). However, Prado (2000) recognized PF as a vegetation unit more closely related to other tropical seasonal forests of South America, owing to its characteristic flora and physiognomy. SSDF covers approximately 10,000 km^2 and shows a broad range of rainfall. Annual rainfall averages 625 mm (range: 450–700 mm) at CF and 820 mm (range: 550–1400 mm) at PF, concentrated during the summer (November to March) (Bianchi and Yañez, 1992). The mean annual temperature 21.5° C is relatively homogeneous, although thermal amplitude is variable in the study area (Arias and Bianchi, 1996). Based on this climatic variability and the common origin of these forests, our main objectives were to identify the tree-species compositional gradient across SSDF and to describe forest structure in different tree communities at the regional scale.

We established 23 1-ha permanent plots, three plots in CF and 20 plots in PF. All plots were 20 m x 500 m corrected for slope, to actually cover 1 ha. A full inventory was made of all trees ≥10 cm in diameter at breast height (dbh). Trees were marked, measured for dbh and height, and identified to species level. For data analysis we performed a Detrended Correspondence Analysis (DCA). DCA is an unconstrained ordination analysis that provides the basic overview of the compositional gradients in species-abundance data (Lepš and Šmilauer, 2003). We calculated basal area, species richness and canopy height in each plot to compare forest structure between sample plots.

We identified 10,029 trees belonging to 116 species, 93 genera and 43 families. The length of the first axis provided an estimate of the high beta diversity in tree species composition in SSDF (5.3 SD units). First and second axes explained about 30% of total species variability (axis 1, 19.8; axis 2, 9.8), whereas the remaining axes explained much less. Distribution of samples and species in a bi-plot suggested that there are three groups or community assemblies at the regional scale. First, near zero on axis 1, were plots located in CF with *Ziziphus mistol*, *Ruprechtia triflora* and *Geoffrea decorticans* as exclusive species. Then, on the other side, upward, were PF plots located toward the west of the study region, and downward were PF plots located toward the east. Rainfall is higher toward the east of the study area, where species such as *Pisonia ambigua*, *Chrysophyllum gonocarpum* and *Diatenopteryx sorbifolia* were abundant. These species are common at higher elevation (i.e. Yungas forest), whereas they are poorly represented in PF plots located in the west. On the other hand, species such as *Ceiba insignis*, *Phyllostilon rhamnoides*, *Calycophyllum multiflorum* and *Astronium urundeuva* were abundant in PF plots located in the west, where moisture stress could be higher because temperatures reach more than 40°C in the summer (Brown *et al*., 2001). When we compared forest structure, mean CF basal area (15.3±2.7 m^2/ha) and canopy height (13.3±3.2 m) yielded lower than average PF values (west PF plots: 22.1±1.7 m^2/ha and 20.6±1 m; east PF plots: 21.3±0.8 m^2/ha and 19.7±0.7 m). Species richness per 1 ha plot varied between all groups: 20 species (range: 18–24) at CF, 35.2 species (range: 22–48) in the west PF, and 39.3 species (range: 32–45) in the east PF.

Our results showed important gradients in species diversity and forest structure in the SSDF, particularly in terms of species richness. Distribution of tree species across and within Yungas forest is strongly affected by climatic factors (Malizia, 2004). We believe that variability in rainfall and temperature could be playing a mayor role in tree species distribution at SSDF in northwestern Argentina. Therefore, an important goal for ecologists in the future is to predict how these forests will respond to climate change.

Box 4.3 (cont.)

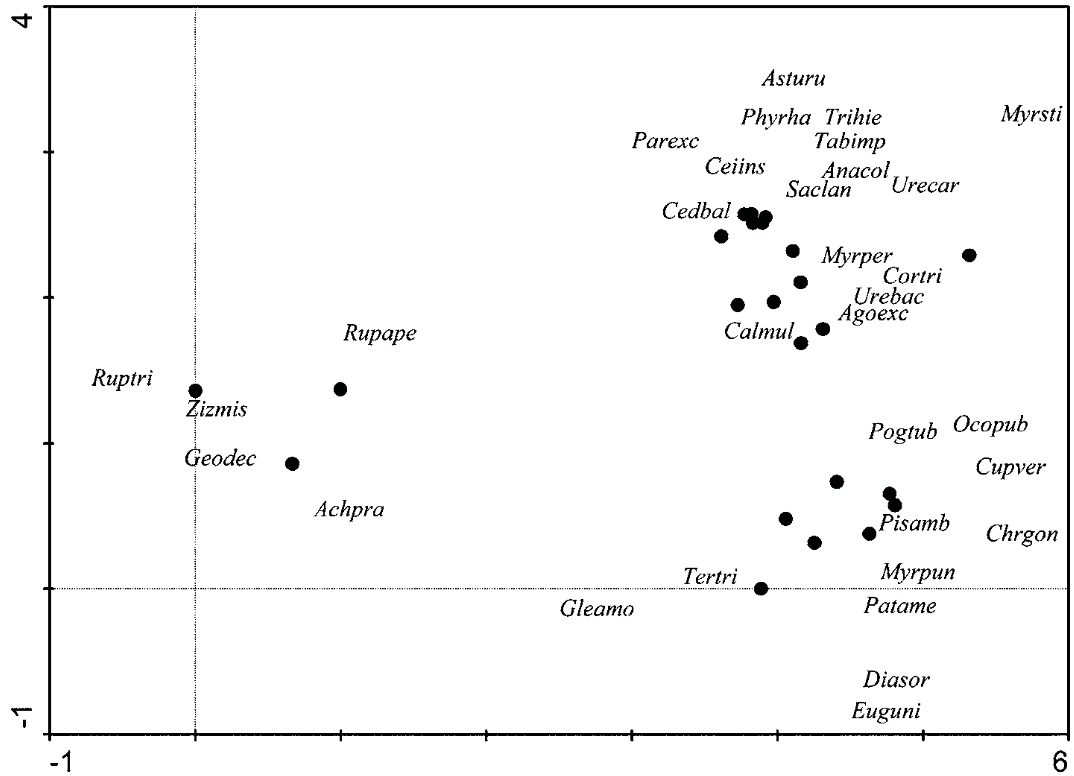

Figure 1 Bi-plot of samples and selected species (>10% weight; 32 species) in DCA. Species codes: Achpra: *Achatocarpus praecox*, Agoexc: *Agonandra excelsa*, Anacol: *Anadenanthera colubrina*, Asturu: *Astronium urundeuva*, Calmul: *Calycophyllum multiflorum*, Cedbal: *Cedrela balansae*, Ceiins: *Ceiba insignis*, Chrgon: *Chrysophyllum gonocarpum*, Cortri: *Cordia trichotoma*, Cupver: *Cupania vernalis*, Diasor: *Diatenopterix sorbifolia*, Euguni: *Eugenia uniflora*, Geodec: *Geoffrea decorticans*, Gleamo: *Gleditzia amorphoides*, Myrper: *Myroxylon peruiferum*, Myrpun: *Myrcianthes pungens*, Myrsti: *Myriocarpa stipitata*, Ocopub: *Ocotea puberula*, Parexc: *Parapiptadenia excelsa*, Patame: *Patagonula americana*, Phyrha: *Phyllostilon rhamnoides*, Pisamb: *Pisonia ambigua*, Pogtub: *Pogonopus tubulosus*, Rupape: *Ruprechtia apetala*, Ruptri: *Ruprechtia triflora*, Saclan: *Saccelium lanceolatum*, Tabimp: *Tabebuia impetiginosa*, Tertri: *Terminalia triflora*, Trihie: *Trichilia hieronymi*, Urebac: *Urera baccifera*, Urecar: *Urera caracasana*, Zizmis: *Zizipus mistol*.

Box 4.4 Patterns of diversity of fungi in an altitudinal gradient

S. Vázquez Mendosa, R.F. del Castillo, R, Valenzuela Garza

Macromycetes are one of the most diverse group of organisms on Earth. However, very little is known about their ecology and species composition, particularly in the tropics. We estimated macromycete richness and productivity in four localities ranging from 2100 to 3120 m elevation in Santa Catarina Ixtepeji, Sierra Madre de Oaxaca, Mexico. In each locality an area of 875 m^2 distributed in 125 plots was surveyed. A total of 1762 specimens were collected, from which 669 morphospecies could be identified. Productivity and species richness were positively correlated with altitude. This pattern was also found when individual groups of fungi were analyzed separately, namely mycorrhizal, saprobic and coprophilous. As ground moisture was also positively correlated with elevation, humidity – more than temperature – appears to be a major limiting factor for macromycetes at the altitudinal range studied. Lignicolous fungi, however, displayed a monotonic relationship with altitude showing a peak of diversity and productivity near 2250 m. This group had the highest biomass in all the studied localities and, with some exceptions, appears to be less affected by humidity than other groups of fungi. Species similarity calculated with Sorensen (presence/absence) and Renkonnen (abundance) indices were very low among localities, suggesting a high species turnover. Conservation and restoration efforts should pay particular attention to middle and high altitude areas in this mountain range, given its high species richness and the vulnerability of its biota because of climate change.

Box 4.5 Species diversity in northwestern Patagonian dryland forests: implications for restoration

C. P. Souto, K. Heinemann, T. Kitzberger, A. C. Premoli

Drylands comprise 30% of the Earth's surface; in South America 94% of Patagonia is exposed to some degree of desertification risk. Particularly in northwestern Patagonia, Argentina, treeless areas have been traditionally viewed by foresters and land managers as barren lands unable to support native forest and have been used for extensive sheep and cattle ranching, or the establishment of exotic (mostly pine) plantations, which drastically impact on dry native forests. Nevertheless, at the dry eastern edge of temperate forest, the endemic conifer *Austrocedrus chilensis* (D. Don) Florin & Boutelje (Cupressaceae) occurs in patches as almost the only tree, while towards the west, this species forms mixed continuous forests with *Nothofagus* species. *Austrocedrus* is considered the most drought-tolerant tree species in the Patagonian region. Particularly towards its driest distributional range, seedling establishment is dependent on shrub presence, since the limiting factors for seedling establishment in open woodland habitats appear to be the desiccating effects of open sites (Kitzberger, 1995; Kitzberger *et al*., 2000), a phenomenon known as 'nurse syndrome'. This syndrome is characterized by the amelioration of micro-environmental variables under a plant – the nurse – which enhances survivorship and/or growth of other species growing in association with it (Raffaele and Veblen, 1998).

We analyzed plant diversity at the landscape scale under heterogeneous and disturbed environments inhabited by *Austrocedrus* forests. The aim of the research was to examine stand structure and species composition, and thus available nurse shrubs along *Austrocedrus'* range, with the aim of informing restoration practices. *Austrocedrus* is a timber species of high economic value and international conservation concern (status: Vulnerable). It is the most conspicuous tree species in the Patagonian steppe. *Austrocedrus* is a dioecious species of a monotypic

Box 4.5 (cont.)

genus, with wind dispersed pollen and winged seeds. In Argentina, it occurs discontinuously from 36°30' to 39°30'S and more extensively from 39°30' to 43°35'S latitude (Seibert, 1982). Over the natural range of the species, most of the precipitation occurs during autumn and winter generating a period of drought in summer. Toward the more humid environments of the south and the west, *Austrocedrus* forms pure stands that tend to be continuous, with a dense understorey of shrubs and other sub-canopy trees. At the centre of its range, the rain shadow effect of the Andes depicts a more evident west to east natural fragmentation gradient. In northern and eastern dry areas, precipitation declines, aridity increases, the understorey of shrubs and small trees becomes less dense, and *Austrocedrus* stands open up into sparse woodlands adjacent to the Patagonian steppe (Seibert, 1982). Finally, scattered trees, typically on rocky outcrops, occur sparsely as ingressions into the Patagonian steppe, surrounded by a matrix of bunch grasses and low shrubs.

To identify the broad-scale trends of differentiation throughout the species range in Argentina, we subdivided sampled stands into three regions representing north (N), centre (C) and south (S), according to their geographical proximity, and environmental envelope. We assessed the presence of all plant species and scored 15 *Austrocedrus* trees along an approximately 20 x 100 m long strip following the major axis in 67 *Austrocedrus*-dominated forest patches on the eastern slopes of the Patagonian Andes in Argentina. Tree ages are significantly higher in fragmented woodlands adjacent to the steppe in the northern and central region of the species range (mean ages are 106 ± 69 and 98 ± 46 years respectively) than in continuous forests occurring in the central and southern regions (52 ±27 and 48 ±23 years respectively). On the other hand, the annual radial growth rate increases significantly from north to south (1.9, 2.3 and 3 mm/year on average in northern, central and southern forest stands respectively), without significant differences in tree diameter (mean tree diameter sizes decrease from 41 in northern forest patches to 38 and 32 cm in central and southern stands). Comparing both latitudinal range extremes, trees from the northern region are older but smaller in size and thus demonstrate a relatively low growth rate.

A total of 89 understory species were recorded in *Austrocedrus* stands (**Table 1**). Only eight species were introduced weeds while naturally occurring plants consisted of 38 herb, 37 shrub, and eight tree species. In northern *Austrocedrus* stands a total of 55 different species were scored in 23 sites (average = 9.39; SD = 7.71), including seven exotic and 14 nurse species. In the central region, 63 species were scored in 25 sites (average = 12.48; SD = 7.39), including three exotic species and 14 nurse species. Meanwhile, in the south 67 species were recorded in 19 sites (average = 15.11; SD = 6.97), including five exotic species and 18 nurse species (**Table 1**). Species richness differed among sampled regions ($F(1,63) = 3.205$, $p = 0.047$). Specifically, tree species richness increased southward. Along the range of *Austrocedrus* in Argentina the three regions shared almost 35% of the understory species, but more than 30% of them were exclusive (i.e. only present in one region). Thus, the C region shared with N and S more than 10% of the species but, N and S only shared 4%. Consistently, the C region has only 8% of exclusive species, meanwhile N and S have almost 20% of exclusive species, respectively. In terms of nurse species, a total of 24 were scored in *Austrocedrus*-dominated communities, which differed across *Austrocedrus'* range in Argentina. The three regions shared 40 % of nurse species. In contrast, 12.5%, 8.33% and 20.8% of them were exclusive from N, C, and S region, respectively.

In summary, species richness and stand structure of *Austrocedrus* vary at the landscape scale, probably in response to climatic and disturbance gradients. Considering scenarios of increasing inter-annual climatic variability and global warming trends, it is possible that climate changes in Patagonia will affect *Austrocedrus* forests. As a consequence, entire regions may change in terms of landscape and forest patch configuration. For successful restoration actions in Patagonian dry lands with *Austrocedrus*, special concern should be given to the presence of nurse species (Chapter 5). The significant genetic structure of *Austrocedrus* (Chapter 7) along with the heterogeneity in community structure and composition reported here should also be taken into account.

Box 4.5 (cont.)

Table1 Plant species diversity from southern Argentina in *Austrocedrus chilensis* stands. Nurse species in bold.

Latitude	*Longitude*	*R*	*Species*
37°58'55.5" to 39°31'26.5"	70°47'19.1" to 70°58'09.6"	N	*Acaena ovalifolia*; *Acaena pinnatifida*; *Acaena splendens*; *Adesmia boronioides*; *Alstroemeria aurea*; *Anemone multifida*; *Araucaria araucana*; *Armeria maritima*; *Baccharis sp.*; *Balbisia gracilis*; ***Berberis buxifolia***; *Berberis empetrifolia*; *Bowlesia tropaeolifolia*; *Cerastium arvense*; *Chusquea couleou*; ***Colliguaja integerrima***; *Cortaderia araucana*; ***Discaria articulata***; *Echium vulgare* (exot); ***Ephedra breana***; ***Ephedra frustillata***; *Escallonia virgata*; ***Fabiana imbricata***; *Gaultheria sp.*; *Geranium magellanicum*; *Haplopappus glutinosus*; *Loasa bergii*; *Lomatia hirsuta*; *Maytenus boaria*; *Maytenus chubutensis*; *Melilotus alba* (exot); *Mulinum echinus*; ***Mulinum spinosum***; *Mutisia sp.*; *Nothofagus antarctica*; *Osmorhiza chilensis*; *Oxalis adenophylla*; *Perezia recurvata*; *Phacelia secunda*; *Plantago lanceolata* (exot); *Quinchamalium chilense*; *Ribes cucullatum*; *Ribes magellanicum*; *Rhodophiala mendocina*; *Rosa rubiginosa* (exot); *Rumex acetosella* (exot); ***Schinus odonellii***; ***Schinus patagonicus***; *Senecio sp.*; *Sisyrinchium vulgare*; *Taraxacum medicinale* (exot); *Tropaeolum incisum*; *Verbascum thapsus* (exot); *Vicia nigricans*; *Viola sp.*
40°43'18.6" to 41°13'25.9"	71°08'27.6" to 70°42'00.8"	C	*Acaena ovalifolia*; *Acaena pinnatifida*; *Acaena splendens*; *Adesmia boronioides*; *Adesmia afin volckmanni*; *Alstroemeria aurea*; *Anemone multifida*; ***Aristotelia chilensis***; *Armeria maritima*; ***Baccharis sp.***; *Balbisia gracilis*; ***Berberis buxifolia***; *Caiophora sp.*; *Calceolaria sp.*; *Cerastium arvense*; *Cynanchum descolei*; ***Colletia hystrix***; ***Discaria articulata***; *Embothrium coccineum*; *Escallonia rubra*; ***Ephedra chilensis***; *Eringium paniculatum*; *Euphorbia sp.*; ***Fabiana imbricata***; *Fragaria chilensis*; *Galium hypocarpium*; *Gaultheria sp.*; *Geranium magellanicum*; *Grisebachiella hieronymi*; *Haplopappus glutinosus*; *Lathyrus sp.*; *Loasa bergii*; *Lomatia hirsuta*; *Maytenus boaria*; *Maytenus chubutensis*; *Muehlenbeckia hastulata*; *Mulinum echinus*; ***Mulinum sponosum***; *Mutisia sp.*; *Myoschilos oblongum*; *Nardophyllum obtusifolium*; ***Nassauvia glomerulosa***; *Orquidea sp.*; *Osmorhiza chilensis*; *Oxalis adenophylla*; *Perezia recurvata*; *Phacelia secunda*; *Quinchamalium chilense*; *Ribes cucullatum*; *Ribes magellanicum*; *Rosa rubiginosa* (exot); *Rumex acetosella* (exot); *Rumohra adiantiformis*; ***Schinus odonellii***; ***Schinus patagonicus***; *Senecio sp.*; *Sisyrinchium vulgare*; *Solidago chilensis*; *Tropaeolum incisum*; *Valeriana sp.*; *Verbascum thapsus* (exot); *Vicia nigricans*; *Viola sp.*
41°47'48.2" to 43°43'24.9"	71°25'51.7" to 71°24'16.4"	S	*Acaena ovalifolia*; *Acaena pinnatifida*; *Acaena splendens*; *Adesmia sp.*; ***Anartrhophyllum strigulipetalum***; *Anemone multifida*; *Aristotelia chilensis*; *Armeria maritima*; *Azorella monantha*; ***Baccharis sp.***; *Balbisia gracilis*; ***Berberis buxifolia***; *Berberis empetrifolia*; *Blechnum magellanicum*; *Bowlesia tropaeolifolia*; *Caiophora sp.*; *Calceolaria sp.*; *Cerastium arvense*; *Cirsium vulgare*; ***Colletia hystrix***; ***Diostea juncea***; ***Discaria articulata***; ***Discaria chacaye***; ***Discaria trinervis***; *Embothrium coccineum*; ***Ephedra chilensis***; *Eringium paniculatum*; *Escallonia rubra*; *Euphorbia sp.*; ***Fabiana imbricata***; *Fragaria chilensis*; *Galium hypocarpium*; *Gaultheria sp.*; *Geranium magellanicum*; *Grisebachiella hieronymi*; *Haplopappus glutinosus*; *Juniperus communis* (exot); *Lathyrus sp.*; *Lomatia hirsuta*; *Maytenus boaria*; *Maytenus chubutensis*; ***Mulinum spinosum***; *Mutisia sp.*; *Myoschilos oblongum*; *Nardophyllum obtusifolium*; *Nothofagus antarctica*; *Nothofagus pumilio*; *Orquidea sp.*; *Osmorhiza chilensis*; *Ovidia andina*; *Oxalis sp.*; *Perezia recurvata*; *Phacelia secunda*; *Plantago lanceolata* (exot); *Polystichum plicatum*; *Quinchamalium chilense*; *Ribes cucullatum*; *Ribes magellanicum*; *Rosa rubiginosa* (exot); *Rumex acetosella* (exot); *Rumohra adiantiformis*; ***Schinus odonellii***; ***Schinus patagonicus***; *Senecio sp.*; *Valeriana sp.*; *Verbascum thapsus* (exot); *Viola sp.*

Box 4.6 Avian-generated seed rain and germination in the patchy shrubland of central Chile

S. Reid, C. Christophers, J.L. Allendes, J.J. Armesto

The pattern of seed rain across a landscape can determine the distribution of potential recruitment rates, influencing the spatial processes of colonization, range expansion and gene flow. Likewise, the quantification of propagule input and germination response in different micro-sites is of fundamental importance to the restoration of degraded plant communities (e.g. Méndez *et al*., 2008). This information is key for understanding the limiting factors for natural regeneration and succession, and for applying information about seed dispersal and germination in restoration programmes.

In this study, we provide evidence of the positive contribution of avian-frugivores to seed dispersal and the outcome of seeds in dispersed sites in the patchy Mediterranean-type shrubland in central Chile. Here, the regeneration of woody species is limited by seed inputs because soil or aerial seed banks of woody species are extremely poor or entirely absent (Fuentes *et al.,* 1984). In addition, regeneration is severely limited to wet micro-sites under bushes in a mosaic of sparse shrub clumps separated by open areas exposed to drought and herbivory by feral rabbits (introduced from Europe), cattle and horses. This shrubland presents a high incidence of avian-dispersed woody species where 50% of the species are dependent mostly on birds for seed dispersal and germination. In addition, seed germination of woody species is not responsive to fire or smoke stimulation. Given the patch structure of Chilean semi-arid ecosystem, we assessed the effect of the present patch structure on avian-generated seed rain and germination.

We quantified avian-generated seed rain patterns directly by collecting defecated seeds from the ground. We searched for avian droppings along five 90 m linear transects separated by 10 m, covering a total area of 450 m^2. Avian-generated seed deposition patterns were based on a total sample of 370 seeds of *Schinus polygamus* (Anacardiaceae) from 95 bird droppings, which made up 80% of the seeds present in droppings. Avian-generated seed rain was compared among different patch-types defined as: (1) 'open', bare ground between shrub clumps; (2) 'low', beneath *Baccharis* sp. and *Retanilla trinervia* (25% shrub cover, 1.5 m mean height); (3) 'midheight', including *Lithrea caustica, Schinus polygamus*, *Azara dentata* and *Colliguaja odorifera* (77.8% mean shrub cover and 3.4 m mean height); and (4) 'tall' (64.6% mean cover and 5.8 m mean height), including *Maytenus boaria* and *Quillaja saponaria*. Germination trials were conducted with *Lithrea caustica* (Anacardiaceae) in the same four contrasting patch types characteristic of this shrubland. In each of the four patches, 60 seeds were placed in open-ended plastic cups (10 seeds per cup) and these were covered with a 5.8 mm mesh cage (50 x 50 cm) to keep out vertebrate herbivores. Six replicates were set up in each patch type, except in the open ground patches where we set up three replicates. The experiment began in the austral winter (August 2006) and germination (emergence of the radicle) was recorded weekly until December (14 records), when germination ceased because of lack of rainfall.

The avian-generated seed rain differed among patch types and differences did not correspond to those expected from the frequency of different patch types in the study area (c^2 = 28.1, df = 3, p <0.001; **Fig. 1**). Dispersed seeds were highly concentrated under 'tall' patches (under *Q. saponaria* and *M. boaria* trees). Tall patches received more seeds than expected based on their ground cover percentage, in contrast to 'open' patches that received significantly lower seed rain than expected. There was also a significant effect of patch type on the cumulative percentage of germination of *L. caustica* (χ^2 = 25.94, p <0.001; **Fig. 2**). Germination was higher under tall and midheight patches and significantly lower under low *Baccharis* patches (Z = 4.95, p <0.001 between low and midheight patches, and Z = 4.12, p <0.001 between low and tall patches). Seed germination was zero in open ground between patches.

Box 4.6 (cont.)

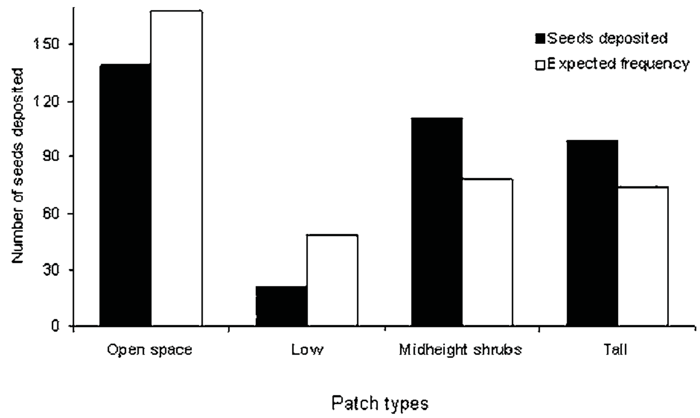

Figure 1 Avian-generated seed rain in different patch types compared to that expected from their observed frequency in the area. Data are compared for *Schinus polygamus* seeds contained in the droppings of birds in a flat landscape of central Chile.

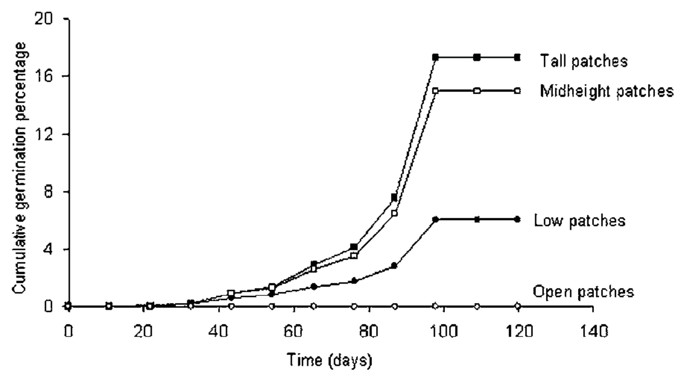

Figure 2 Cumulative germination percentage of *Lithrea caustica* under four different patch types: tall (■), midheight (□), low (●) and open (○).

In this study, we showed that the majority of seeds (62% of 370 seeds) transported by birds are dispersed to sites were germination and seedling survival have higher probabilities of success, although a large percent is still also dispersed to open areas. Considering seed germination rates were higher under tall and midheight patches of *Q. saponaria* and *L. caustica* respectively, we suggest that dispersal to these 'nurse' patches is positive for plant recruitment, in agreement with previous studies in this shrubland (e.g. Del Pozo *et al.*, 1989) and Mediterranean shrublands in southern France (Debussche and Isenmann, 1994). Consequently, avian-frugivores are contributing 'direct' seeds to favourable sites for germination. This disperser-mediated facilitation which leads to the aggregation of seedlings of woody plants around pre-existing shrub 'nurses', offers a tool for restoration, as it can accelerate succession, by mostly driving the early appearance of late-successional species (e.g. *M. boaria, L. caustica,* and *Schinus* sp. in this shrubland). Most likely succession could slowly lead to the coalescence of patches from seedling recruitment along patch edges, but this requires decreased herbivory and high moisture conditions.

We recommend the conservation of tall patches (e.g. *Q. saponaria* and *M. boaria*) as important to facilitate succession in shrublands of central Chile. This guideline can be applied in restoration practices in degraded shrubland or dry forest areas, by planting or protecting remnant adult *Q. saponaria* and *M. boaria* to facilitate progagule input. Protection from herbivory should also be necessary to accelerate the process of seedling establishment under patches.

Box 4.7 Effect of fragmentation on plant communities of central Chile

P. Becerra, C. Smith-Ramírez, C. Echeverría, J.J. Armesto

We assessed the consequences of fragmentation for plant species diversity and population densities of remnant sclerophyllous forests of central Chile. We selected 41 woodland fragments of different patch sizes (1 to 2000 ha). Each fragment was characterized by several fragmentation variables related to area and shape (area, core area, shape index, fractal dimension). In addition, we quantified the distance from each forest patch sampled to the nearest urban centre and recorded fire, and also the elevation of the sites where each fragment was sampled. We tested the hypotheses that different communities and population variables vary along gradient of fragmentation, distance to urban centres and fires, and elevation.

We found that larger fragments were also more complex in shape. Only some community and population variables were significantly related to fragmentation. Tree, shrub and epiphyte, but not herb species richness, and diversity increased in larger and more complex fragments. In turn, only tree species richness was significantly correlated to the distance to urban centres and no community variable to distance to fires. Also, density of different size classes of particular tree species were significantly correlated with fragmentation variables, although with different patterns observed for different species. Density of more shade-tolerant species was positively correlated to patch area and complexity, while density of more shade-intolerant tree species was not. Also, density of old individuals of two tree species were positively correlated with distance to fires suggesting that fires could have negatively affected density of some tree species. Finally, elevation was significantly positively correlated with richness, diversity and density of some species suggesting that climatic variation with elevation affects plant communities and populations. In conclusion, patterns detected in this study suggest that fragmentation and elevation are relevant factors affecting communities and populations of plant species in this Mediterranean-climate ecosystem.

Details of methods

Field surveys were undertaken in forest fragments classified into four size classes, to ensure that a broad range of patch sizes, representative of remnant woodlands in the landscape, was sampled. Four classes were: 0.5–10 ha, 10–100 ha, 100–1000 ha, >1000 ha. From each size area we selected 22, 9, 7 and 3 patches respectively. Thus we worked with 41 fragments distributed between 32°S and 34°S, located only on the coastal range of central Chile. Surveys were undertaken by establishing one 10 x 10 m plot per patch in the two smaller patch size classes, and two plots of the same dimension at least 300 m apart in fragments belonging to the two larger patch size classes. Within each sample plot we recorded the dbh of each individual >5 cm of all tree species. In addition, we sampled 15, 1 x 1 m sub-plots located systematically within the larger plot. In each sub-plot we recorded the presence of each vascular plant species, including herbs and ferns, and the regeneration (seedlings and juveniles) of tree species. Regeneration was subdivided in two classes: saplings including individuals <5 cm dbh higher than 0.5 m, and seedlings including individuals <0.5 m height. Additionally, evidence of recent fires (yes or no), recent logging (yes or no) and/or recent trampling or browsing by livestock (faeces) (yes or no) was noted for each plot and the 20 m around it.

Box 4.7 (cont.)

Table 1 Statistical results of simple correlations between environmental or fragmentation variables and plant variables included in the study. R values of correlations are shown and significant values (p <0.05) are shown in bold. Dependent variables for exotic species consider all life forms together. Total density of regeneration corresponds to the sum of individuals in seedling and sapling stages. From Becerra *et al.* (2010).

	Distance to urban centres	Distance to fires	Area	Shape index	Fractal index	Core area	Elevation
Cover 1–2 m (%)	0.04	0.24	**0.45**	**0.60**	**0.53**	**0.38**	0.23
Richness tree species	**0.31**	−0.08	0.17	**0.37**	**0.42**	0.12	0.28
Diversity tree species	0.27	−0.18	0.19	**0.36**	**0.38**	0.13	0.28
Richness shrub species	−0.24	−0.10	0.17	**0.33**	**0.33**	0.11	**0.31**
Richness epiphyte species	0.29	−0.17	**0.33**	**0.41**	**0.46**	**0.29**	**0.52**
Diversity epiphyte species	0.29	−0.23	0.23	**0.31**	**0.36**	0.21	**0.42**
Richness seedlings	0.26	−0.17	0.10	**0.37**	**0.50**	0.05	**0.38**
Richness saplings	0.13	0.06	0.27	**0.49**	**0.53**	0.20	**0.44**
Richness class 5–15	0.20	−0.24	0.25	**0.34**	**0.31**	0.20	0.10
Density saplings	0.12	−0.02	**0.60**	**0.61**	**0.51**	**0.54**	0.18
Density class 5–15	**0.40**	−0.05	**0.30**	0.15	0.16	**0.34**	0.13
Density class 15–30	0.17	**0.32**	**0.31**	0.16	0.17	**0.39**	−0.09

Box 4.8 Tree species diversity driven by environmental and anthropogenic factors in tropical dry forest fragments of central Veracruz, Mexico

G. Williams-Linera and F. Lorea

We examined vegetation structure and woody species diversity in relation to 14 environmental and anthropogenic factors in ten tropical dry forest (TDF) fragments in central Veracruz, Mexico. The basal area of the canopy (30.2 ± 2.11 m2/ha) and understory (1.96 ± 0.12 m^2/ha) trees was similar, but density (1014 ± 104 and 2532 ± 227 individuals/ha, respectively) differed among sites. We recorded 98 canopy, 77 understory, and 60 seedling species. Richness was 24–45 species per site, Fisher's alpha and Shannon's indices increased with site altitude. Chao Jaccard indices revealed high species turnover, and a consistently higher similarity within the sites at the lowest and within the highest elevation sites. Ordination identified altitude, aspect, slope, water proximity, cattle and trails as significant explanatory variables of species patterns, and showed that sites at lower elevations were clearly separated from the other sites. Environmental heterogeneity alone did not control species diversity distribution, but species were affected by environmental filters at different stages in their life cycle, e.g. water proximity was significant for saplings and seedlings but not for adults. Anthropogenic disturbances act synergistically, e.g. trails played a key role in determining structure and tree diversity patterns. An important finding is that human disturbance diminishes species diversity in this TDF, but sites at lower elevations were more disturbed and less diverse. There is therefore a need to study how environmental factors would act if there were no anthropogenic disturbance. Full details of this study are presented by Williams-Linera and Lorea (2009).

We found no differences between GLIMs with and without interactions among patch attributes and environmental variables as predictors of tree species richness, and therefore we selected models without interactions, and the lower number of parameters for all the analyses presented. We found a significant effect of the study sites and the country on tree species richness patterns, but no effects of patch area and elevation (**Table 4.2**). We also found a significant effect of elevation and site, but not of patch area, on adult tree densities in remnant forest patches from all sites (**Table 4.3**). With regard to species richness of tree seedlings, we found significant effects of elevation, site and country, but not of patch area (**Table 4.4**). Finally, for the variation in tree seedling densities among forest patches, we found no significant effects of environmental or patch variables, and no effects of site or country (**Table 4.5**).

Table 4.2 Effects of elevation, forest fragment area, country, sampling sites (Veracruz, Oaxaca and central Chile) on tree species richness in seasonally dry, tropical (Mexico) and Mediterranean (Chile) forests. Significant model effects p <0.05 are shown with *.

	Estimate	*Standard error*	*T value*	*Probability – ItI*
Elevation	−3.11	1.74	−1.78	0.07
Fragment area	0.85	0.47	1.81	0.073
Country	22.6	2.05	11.0	<0.001*
Sampling site	−6.30	0.78	−8.04	<0.001*

Table 4.3 Effects of forest fragment area, elevation, sampling sites (Veracruz, Oaxaca and central Chile), and country on tree abundance in seasonally dry, tropical (Mexico) and Mediterranean (Chile) forests. Results obtained with a GLIM model, significant effects are asterisked.

	Estimate	*Standard error*	*T value*	*Probability -ItI*
Altitude	0.0004920	0.0001830	2.688	0.009*
Area	−0.0001411	0.0001286	−1.097	0.277
Country	-	-	-	-
Site	0.5517715	0.1830766	3.014	0.004*

Table 4.4 Effects of forest fragment area, altitude, sampling sites (Veracruz, Oaxaca and central Chile), and country on seedling tree species richness in seasonally dry, tropical (Mexico) and Mediterranean (Chile) forests. Results obtained with a GLIM model, significant effects are asterisked.

	Estimate	*Standard error*	*T value*	*Probability -ItI*
Altitude	9.224e−03	2.329e−03	3.960	<0.001*
Area	4.737e−04	6.211e−04	0.763	0.448
Country	4.587e+01	5.591e+00	8.205	<0.001*
Site	3.329e+01	4.881e+00	−6.820	<0.001*

Table 4.5 Effects of forest fragment area, altitude, sampling sites (Veracruz, Oaxaca and central Chile), and country on seedling tree abundance in seasonally dry, tropical (Mexico) and Mediterranean (Chile) forests. Results obtained with a GLIM model, significant effects are asterisked.

	Estimate	*Standard error*	*T value*	*Probability -ItI*
Altitude	0.0017392	0.0018324	0.949	0.346
Area	0.0007402	0.0004885	1.515	0.135
Country	6.8388384	4.3976176	1.515	0.125
Site	−7.0301651	3.8390390	−1.831	0.072

All of the analyses presented showed a significant or marginally significant (p = 0.05 to 0.1) effect of the study sites on tree species richness. In other words, results were dependent on the specific region of Mexico or Chile where dry forests occurred. Further, elevation was positively and significantly related to tree densities and species richness of tree seedlings in all sites. This means that patches occurring at higher elevations generally presented greater densities of trees and more abundant regeneration, but no significant trends in species richness of adult trees were recorded. At the same time, patches at increased elevation had more species of tree seedlings, but not higher densities of tree seedlings, than lower elevation patches. In all of the analyses in both countries, the area of remnant patches had no effect on the species richness of trees or tree seedlings.

Studies in other forest types have attributed the lower species richness found at higher elevation to environmental stress factors such as harsher climate and infertile soils (Rahbek, 1995; 1997; Bachman *et al.*, 2004; Smith-Ramírez *et al.*, 2007). However, we are not aware of previous reports that documented increasing tree diversity with elevation. Potentially, changes in the duration of the dry season and moisture availability at different elevations could influence the species richness of seedling and adult trees. In Chilean coastal hills, oceanic fog frequently covers the mountaintops above 500–600 m elevation, which consequently receive significantly more precipitation than lowland areas (del-Val *et al.*, 2006). This could positively affect the recruitment and survival of tree species, by increasing habitat and resource heterogeneity. Furthermore, the patterns found for tree species in relation to altitude are not the same as those for herbs and shrubs. In Chilean dry-sclerophyllous forests, greater species richness of bulbs and herbs occurs in lowland areas, and hence, if all plant species (woody and non-woody) are included in the analyses, the highest number of species is found at low elevations (U Chile, 2007).

Because greater anthropogenic impact affects the vegetation of lowland areas in Chile (Armesto *et al.*, 2010), Veracruz (Williams and Lorea, 2009; **Box 4.8**), and probably Chiapas (Neptalí Ramírez-Marcial, personal communication), it is likely that two important factors, altitude and antropogenic impact, are acting together, or one may be masking the effects of the other. For example, in central Chile, the distance from forest fragments to cities and towns is negatively correlated with woody species richness (Becerra *et al.*, unpublished manuscript). Future studies should elucidate the relative importance of both factors.

Conclusions

One of the most important decisions with regard to restoration programmes is where restoration actions should be undertaken to obtain the best results, in terms of recovery rates of native vegetation cover and species richness. Key questions include: is the tree species richness greater in large than in medium or small forest fragments? What is the threshold patch size of remnant fragments that could sustain the highest tree species richness or density of regeneration? What physical factors of an individual site are important to consider when selecting sites for restoration? Our research did not identify a standard answer to these questions that was valid for all dryland forests analyzed. Rather, the effect of site (country and province) was stronger than other effects, highlighting the importance of local context when identifying restoration priorities. Results suggest that restoration is likely to be more successful in terms of impact on species richness when restoration activities are conducted at higher elevation that in lowland areas, and when the size of the remnant fragments is relatively large. However, the conditions of each site must be analyzed separately. We conclude that environmental factors related to variation in altitude and other specific variables associated with disturbance history in each study area are important determinants of the diversity of adult trees and tree seedlings in American dryland forests. Effects derived from local anthropogenic impact must be analyzed separately to fully understand the processes that account for the present patterns of tree species richness in each region.

Deforested dry forest landscape in Chiapas, Mexico. Photo: R. Vaca

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5 EXPERIMENTAL ANALYSIS OF DRYLAND FOREST RESTORATION TECHNIQUES

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Introduction

Forest landscape restoration in dry forest in the Americas is an urgent priority. Tropical, subtropical, and temperate seasonally dry forests in Mexico, Chile, and Argentina are represented by a wide range of different forest types from Mexican tropical dry deciduous forest, Chilean sclerophyllous and deciduous dry forest, Argentinean subtropical seasonally dry forests – including Andean premontane forest – and the transition to dry Chaco forest, to the forest-steppe ecotone on the eastern slopes of the Patagonian Andes. What all these forests have in common is that they are largely threatened by deforestation (Chapter 2), the establishment of plantations of exotic tree species, overharvesting (particularly wood for fuel), conversion to cropland, and cattle-raising. Worldwide, dryland forest ecosystems have been degraded by unsustainable land-use practices that may alter the structure and composition of forest stands, as well as reduce tree density and the extent of canopy cover. These effects can have serious negative impacts on the ecological processes influencing forest dynamics, including seed dispersal, seed germination, seedling establishment, and growth. As a result, the ability of forests to regenerate naturally can be significantly impaired, reducing the viability of forest patches and threatening the provision of environmental services to local communities as well as the biodiversity associated with dryland forest ecosystems. The development of successful restoration approaches depends on understanding the capacity of forest regeneration and how different human activities influence it.

Ecological restoration is increasingly adopted as an approach to land use in areas that have suffered ecological degradation as a result of human impact (e.g. Lindenmayer and Franklin, 2002; Lamb and Gilmour, 2003; Mansourian *et al.*, 2005). Rey Benayas *et al.* (2009) summarized those types of human activity resulting in degraded ecosystems and the forms of restoration action that are currently being undertaken to address this problem. Restoration actions typically focus on the reduction or removal of factors causing environmental degradation and/or the re-establishment of key ecosystem components to influence the rate and direction of recovery. The simplest approach is passive restoration or cessation of the causal action, accompanied by recovery through natural processes, while other active measures include activities such as tree planting (Rey Benayas *et al.*, 2009).

The development of successful restoration approaches depends in part upon understanding the capacity of forest stands to regenerate naturally and how this process is influenced by different human activities. The main approaches used for forest restoration include encouraging natural regeneration and artificially establishing trees within or around degraded forest stands. In situations where forests are so degraded that natural regeneration is inadequate, such artificial establishment methods may be preferable. A successful outcome from both approaches depends on the understanding of successional processes in dryland forest communities (Quesada *et al.*, 2009) and the factors influencing the establishment and growth of tree seedlings in degraded sites.

Recent research on drylands has focused on investigating restoration from natural regeneration (Aronson *et al.*, 2005). The factors that influence succession must be understood in order to capitalize on natural regeneration mechanisms (Walker *et al.*, 2007). Griscom *et al.* (2009) included in their recommendations for the early stages of forest succession the exclusion of cattle, making site-specific decisions about herbicide application, and the active conservation and protection of riparian zones that function as a critical source of diverse propagules. Vieira and Scariot (2006a) considered the ecology of tropical dry forest regeneration as a tool to restore disturbed lands. Seed collection, planting time, growth of established seedlings, and re-sprouting ability as a prominent mechanism of regeneration in dry forests are factors that must be taken into consideration. With respect to natural regeneration, studies have examined soil seed banks as a source of propagules (Uasuf *et al.*, 2009), seed rain (Ceccon and Hernández, 2009), and seed fate by desiccation or insect predation in abandoned dry forest pastures (Vieira and Scariot, 2006a). However, the number of seed bank species and the quantity of soil-stored seeds are typically relatively low, requiring the natural regeneration process to be assisted through direct seeding, seedling plantation, and the manipulation of a site to improve the environmental conditions for seedling establishment and growth (Uasuf *et al.*, 2009).

Enrichment planting has been assessed as a potential reforestation tool that could complement natural regeneration (Griscom *et al.*, 2005) along with seedling plantation in logged forests (Vieira *et al.*, 2007). Forest succession may be accelerated and planting techniques improved if the use of fertilizer, mycorrhizal inoculation, irrigation, and herbicides is considered. Obviously, cattle and other livestock must be removed prior to enrichment planting (Gricom *et al.*, 2005, Montagnini, 2005).

Other restoration efforts have studied the growth and productivity of native species in pure and mixed plantations, comparing them with the exotic species that are widely used, such as *Tectona grandis* in Mexico and Central America. There is a consensus that the use of mixed plantations with native species is always preferable because of its contribution to sustainable management; single-species plantations are generally of lower biodiversity value and do not provide as great a range of goods and services as natural forest, although mixed plantations are likely to increase this range of benefits (Piotto *et al.*, 2004, Montagnini, 2005).

Several different approaches have been taken for the restoration of dryland systems. The use of living fence species as a restoration tool has the advantage of planting tree species vegetatively; species can act as seed recruitment foci by attracting seed dispersers and provide shade to improve micro-climatic conditions for seedling establishment (Zahawi, 2005). The use of precipitation pulses is essential for dryland regeneration. The increasing ability to predict El Niño-Southern Oscillation (ENSO) effects can be used to enhance management strategies for the restoration of degraded ecosystems (Holmgren *et al.*, 2001). Since dryland ecosystems are dependent on rainfall pulses for their regeneration, understanding the complex effect of ocean conditions may be critical for their management and ecological restoration (Caso *et al.*, 2007).

Planted *Maytenus spp*. seedling in Chile. Photo: C. Echeverria

Nursery tree in Chile. Photo: C. Echeverria

Selection of species for restoration activities is an issue of vital importance. The preferences of local people should be taken into account when planning restoration (see Chapter 5; Montagnini, 2005; Suarez *et al.,* submitted). Garibaldi and Turner (2004) suggested that we restore not only landscapes but also the diversity-enhancing capabilities of the human communities inhabiting those landscapes. They proposed the identification of cultural keystone species that play more than one role; often these roles are supported and enabled by other non-keystone species. Cultural keystone species may play a paramount role in restoration. In addition, restoring forest remnants may increase their valorization by local land users. A non-random pattern of forest degradation has been identified, and there is a risk of potential loss of the most degraded forest remnants unless active forest restoration plans are applied (Tarrasón *et al.,* 2010).

This chapter describes a series of field experiments and surveys that were undertaken in order to identify the main constraints to the establishment and growth of tree species in degraded forest stands, and to determine how these constraints may be overcome through practical management interventions (see also **Boxes 5.1–5.11** for associated investigations). The experiments examined natural and artificial establishment of tree species subjected to a variety of different management approaches, with emphasis on tree species of conservation and/or socioeconomic importance to improve the value of forest resources for local communities. The objectives of this research were as follows: (1) to test forest restoration and land reclamation techniques for reversing degradation and loss of dryland forest ecosystems through a programme of field surveys and experiments established in the study areas, (2) to identify the key ecological processes limiting establishment and growth of threatened and/or socioeconomically important native tree species on degraded forest sites, and (3) to identify appropriate methods of restoring dryland forest ecosystems that contribute both to the conservation and restoration of biodiversity and the economic development of local communities. This chapter presents the results of the programme of field experiments that were established in each study area, and identifies the key ecological processes limiting establishment and growth of threatened and/or socioeconomically important native tree species on degraded forest sites. A discussion is then presented of the implications of these results for the identification of appropriate methods for restoring dryland forest ecosystems that contribute to the conservation and restoration of biodiversity as well as to the economic development of local communities.

In the ReForLan project, two practical approaches to forest restoration were studied: (i) encouragement of natural regeneration, and (ii) artificial establishment of tree species within degraded/deforested stands. Various complementary restoration experiments were conducted in six dryland regions in Latin America, which permit a broad overview of the challenges that practical ecological restoration must confront. Other studies permit exploration of aspects related to active and passive restoration in drylands. Experimental approaches differed between the study areas in accordance with contrasting local circumstances. Together, the different approaches presented here are key to identifying solutions to the problem of degraded drylands in Latin America. The experimental areas were located in three countries: Chile (Central Valley and Coastal Range), Argentina (northwestern and Patagonia), and Mexico (Veracruz and Oaxaca). Other studies relating to restoration activities (not on plantations) were carried out in some of the areas mentioned above and in Chiapas, Mexico. General characteristics of each region are presented in Chapter 1.

Box 5.1 Holistic ranching and landscape restoration in Chiapas, Mexico

B. G. Ferguson and R. Alfaro Arguello

The widespread conversion of tropical forest to pasture was perhaps the most significant land use change in Latin America during the second half of the twentieth century (Kaimowitz, 1996). Forest restoration therefore typically takes place within landscapes dominated by cattle ranching and its success depends upon its compatibility with ranch management. However cattle ranching in the American tropics has been based largely on grass monocultures, a production model that threatens tree cover at the pasture and landscape scales through its inefficient, extensive nature and dependence upon fire and herbicides (Villafuerte *et al*., 1997; Savory *et al*., 1999; Sánchez *et al*., 2003; Szott *et al.*, 2000, Roman-Cuesta *et al*., 2003; Vieira and Scariot; 2006b). Soil and pasture degradation and biodiversity loss increase dependence on herbicides, pesticides, fertilizers and feed supplements, reducing profit margins and driving pasture expansion.

To break this vicious circle, a small group of ranchers in the Central Valley of Chiapas turned to the holistic management (HM) decision-making framework (Savory *et al*., 1999). HM focuses upon relationships among the land, people and their communities and is designed to confront the challenges of land management where humidity is markedly uneven throughout the year, as in areas of tropical dry forest. In 1994, these ten ranchers formed an 'Intensive, Technical Grazing' club ('PIT Las Villas'). In 2007 we documented the advances of the seven remaining members by comparing their management, plant communities and soils to those of 14 neighbouring 'conventional' ranches (Alfaro Arguello, 2008; Ferguson *et al*., in review).

Key management elements we observed on each of the holistic ranches include:

- Numerous pasture divisions created using electric and living fences;
- Frequent rotation (at least daily) of cattle among pasture divisions, maintaining high stocking density and adequate recovery periods;
- Complete or almost complete elimination of use of fire, herbicides, pesticides and chemical fertilizers;
- Manual weeding eliminating only plants that cattle do not consume or that might scratch their udders;
- Maintenance of forest reserves several ha in extent;
- Diversification of forage resources with trees and shrubs such as *Guazuma ulmifolia*, *Pithecellobium dulce* and *Enterolobium cyclocarpum*, grasses and herbaceous legumes;
- Diversification of ranch products (e.g. pigs, sheep, poultry and honey, breeding stock and semen, and timber from rotational harvests);
- Good treatment and adequate spaces and installations for livestock;
- Commitment to constant experimentation, learning and rancher-to-rancher capacity building; and
- A planning process oriented toward quality of life and community well-being as well as to productivity and profits.

In comparison with their conventional neighbours holistic ranches have achieved:

- Greater milk productivity;
- Reduced cow and calf mortality;
- Diminished dependence on purchased inputs, including agrochemicals, feed and hay;

Box 5.1 (cont.)

- Better grass cover;
- Deeper topsoil; and
- Higher soil microbial respiration rates and increased earthworm presence.

We also observed (but did not measure) forest regeneration on the hills surrounding holistic pastures as well as a (non-significant) tendency toward more trees in holistic than conventional pastures. Thus HM boosts ranch productivity at the same time that it fosters tree cover on ranches and surrounding landscapes. HM and other production models based upon agroecological principles are essential elements of ecological restoration of tropical dry forest landscapes. Reforestation efforts that do not address the underlying drivers of deforestation will not succeed beyond the short term. We hope that the experience of PIT Las Villas will encourage other ranchers, large and small, to try HM, and will encourage government support for such strategies. Further investigation will be necessary to identify appropriate technology, training and support mechanisms.

Box 5.2 The role of cattle in tropical dry forest regeneration in Chiapas, Mexico

B. G. Ferguson, M. Rueda Pérez, G. Pascacio Damián, L. Domínguez Morales, P. Bichier

Lack of seed dispersal is one of the most common barriers to regeneration of neotropical forests (Holl, 1999). Intriguingly, cattle, a familiar feature of many neotropical landscapes, can move among vegetation types, consuming and dispersing seeds of woody plants (Janzen and Martin, 1982; Miceli-Méndez *et al*., 2008). Miceli-Méndez and colleagues (2008) identified 13 cattle-dispersed tree and shrub species in Chiapas, mostly in the seasonally dry tropics. They proposed that seed dispersal by livestock holds potential for management as a forest restoration tool, but emphasized the need for a better understanding of the ecology of the phenomenon. We explored three aspects of the role of livestock in dry forest succession: the seed banks present in cattle dung, the role of cattle-dispersed species as nurse trees, and the population structure of one cattle-dispersed species in active pastures.

For the seed bank study, we sieved cattle dung and soil samples from 14 ranches in the Villaflores and Villacorzo municipalities. We found greater diversity and density of tree seeds in manure than in pasture soils. Tree species dominating the manure seed bank included *Guazuma ulmifolia* (Sterculiaceae), *Ficus spp.* (Moraceae) and the legumes *Enterolobium cyclocarpum* and *Acacia spp*. Density of herbaceous seeds, however, was higher in soil than in manure. We suspect that these smaller, softer seeds do not easily survive passage through the bovine digestive tract.

We worked on seven ranches in the same area to quantify seed dispersal and micro-climate beneath two cattle-dispersed tree species (*Guazuma ulmifolia* and *Pithecellobium dulce*) and in open pasture. During a full year of seed trapping with 36 m^2 of traps we recovered 55,832 seeds belonging to 173 morphospecies. For both species, seed rain was significantly greater beneath the canopy than in the open. These differences were particularly marked for tree seeds and for seeds of species dispersed by animals that consume their fruit. Seed rain of half a crown radius outside of tree crowns was not significantly greater than that detected in the open. Micro-climate was significantly cooler and darker under tree crowns. A complementary bird survey recorded 104 species using the ranch landscapes (including pastures, riparian vegetation, fence lines and forest patches), including 30 species observed visiting isolated *G. ulmifolia* and/or *P. dulce*.

We documented the population structure of trees in five pastures near the village of Ocuilapa in the Ocozocoautla municipality. *Acacia pennatula*, a cattle-dispersed species, was present in all five pastures and was by far the most abundant of the 25 tree species we encountered. The

Box 5.2 (cont.)

species accounted for 54% of the 55+33 trees/ha (diameter at breast height >10cm) in our pasture census. In our seedling and sapling plots (diameter at breast height <5 cm), we found 0.93+0.45 individuals/m^2, of which 99% were *A. pennatula*. Ranchers reported at least one use for 80% of the tree species in their pastures. *Acacia pennatula* is used for fodder and shade for livestock, post and construction wood and, most importantly, for fuelwood.

Taken as a whole, these three studies demonstrate that: cattle are effective dispersers of several dry forest tree species that are common in pastures; birds frequently visit these trees within the agricultural landscape; cattle-dispersed trees of modest size act as foci for seed dispersal and as nuclei for tree seedling establishment; and ranchers value and protect some cattle-dispersed trees. In effect, these are autochthonous silvopastoral systems that arise from local rangeland management practices. Understanding the functioning of these systems will contribute to novel strategies for dry forest restoration through livestock management. Furthermore, recognition of these agroforestry systems as such may justify changes in natural resource management policies that sanction and promote these systems as elements of sustainable cultural landscapes.

Figure 1 *Acacia pennatula* seeds (lower right) and seedling in cattle dung on a ranch at Ocuilapa, Chiapas, Mexico. Photo: C. Echeverria

Seedling monitoring in central Veracruz, Mexico. Photo: C. Alvarez

Prehispanic mound in Acazonica, Mexico. The tropical dry forest of central Veracruz has numerous remains of Prehispanic settlements (600 to 1500 AD). Photo: G. Williams-Linera

Box 5.3 Evaluation of commercial plantations and their management in the tropical dry forest region of Paso de Ovejas, Mexico

R.A. Pedraza Pérez

In Mexico, there are two programmes administered by the National Forestry Commission (CONAFOR) to gain access to government incentives and to establish trees on plantations. One of these programmes is 'Pro Árbol' previously known as PRONARE (the Spanish acronym for National Programme of Reforestation 1992–2001), which distributes trees that are propagated in nurseries and sometimes provides other support services relating to reforestation and the maintenance and protection of reforested areas. The other programme is known as PRODEPLAN (the Spanish acronym for Programme for Commercial Plantation Development) that began operations in 1997 and was redesigned in 2001. The main objective of PRODEPLAN is to encourage production of inputs to provide the forest industry with competitive prices in addition to generating jobs and decreasing the stress on natural forests. Such support includes the following aspects of management programmes: establishment, maintenance, insurance premiums, technical support, and development. The plan also finds sources of financing for the cultivation and management of forest species in agricultural lands that have lost native vegetation, leading to the production of timber and non-timber raw materials for commercialization or industrialization. The native species from tropical zones that are most heavily used are mahogany (*Swietenia macrophylla*) and cedar (*Cedrela odorata*); the exotic species are teak (*Tectona grandis*), white beech wood (*Gmelina arborea*), and pink cedar (*Acrocarpus frainifolius*), among others.

The main objective of this study was to determine the number and kind of plantations established in the tropical dry forest of central Veracruz, Mexico based on the official census list. Ten cases were selected in order to ascertain the farmers' purposes; the performance of selected species and established trees was recorded in terms of survival, height, and diameter as well as the commercial wood volume. Ten plantation cases were selected on the basis of the following criteria: (a) tree life form, (b) plantation of at least one hectare, and (c) individual trees planted close together. All trees found on ten 10 x 10 m plots were measured at each site. Empty spaces and stumps were counted and additional information was requested from the landowners. Age and density of plantations were obtained. Survival, total and commercial height, and mean annual increment (MAI) among sites and species were compared.

From 119 recorded cases between the years 2000 and 2006, only 39 met the selection criteria, but they represented 70% of the total planted surface (around 1100 ha). This means that few or scattered trees were planted in most cases, the preferred species being *Casuarina equisetifolia*, *Cedrela odorata*, and *Swetenia macrophylla.* Insufficient tree plantations resulted, forcing us to use cases data from other years to complete ten study cases. The main support from CONAFOR was seedlings grown in their official nurseries, while PRODEPLAN gave 1.5 million pesos to seven plantations established on 208 ha. Some 75% of the total trees studied (684) belonged to *Cedrela odorata*, which is renowned for its natural beauty and outstanding physical properties. That is the reason for their removal from tropical forest and for the species' present endangered status. Their establishment in tree plantations is difficult because of *Hypsipyla grandella* (Zeller) larval attack (Mayhew and Newton, 1998). Tree individuals of *Tabebuia rosea* (13%), *T. donnell-smithii* (9.4%), and *Swetenia macrophylla* (3.5%) were found on three of the plantations studied.

The ten selected sites were very different from each other due to variation in plantation age, tree density, and owners' objectives. Tree density per hectare reflected landowners' objectives upon establishment of the plantation and provided useful criteria for classifying study sites: (1) agro-systems (185 trees/ha) on flat terrain combined with lemon trees (*Citrus limon*); watering and fertilization were permanent on Don Rene (10 ha) and Casa Blanca (16 ha), established three and six years ago, respectively; (2) secondary vegetation under low density tree plantation (216 trees/ha) on the slope of Guaje Mocho (1 ha), established seven years ago; (3) multipurpose mixed plantation with middle and high densities, irrigation frequent, Paso de Varas (556 trees/ha), La Guadalupe (625 trees/ha) and Loma Coyotes (1666 trees/ha), 3, 30, and 4 ha, established three, five, and 12 years

Box 5.3 (cont.)

ago, respectively; (4) conventional density (1111 trees/ha), La Covadonga (six years old) with 62 ha where natural forest was cut before, and Cascajal, La Gloria and Palo Verde (ten years old with 1–5 ha), the three sites were used for agriculture and cattle husbandry. La Covadonga was the only plantation that received financing for its establishment; today it is for sale.

Mean percent survival was high (70%); agro-systems had the highest (100%) survival rate because they replaced dead trees in the first year. Plantations that were ten or more years old had the lowest (16–43%); farmers delayed tree cutting to reduce competition among them. The total height of Mexican cedar increased with plantation age, from 0.89±0.85 m (3 yr) to 6.70±2.4 m (6 yr) and up to 13.3±1.44 m (12 yrs), respectively. Don Rene (3 yr) was 8.3±1.53 m, the product of intense management. In this area, trees did not show evidence of *Hypsipyla* attack. However, all other plantations did, resulting in reduced commercial height. The diameter followed the same tendency as height: the largest measurements were for Don Rene (14±1.3 cm), Casa Blanca (14±2.4 cm), and Guaje Mocho (18±2.8 cm), all with low plantation density. The highest mean annual increment (MAI) for height and diameter was measured on Don Rene (2.8 m and 4.8 cm annually) and Guaje Mocho (1.58 m and 2.6 cm). The commercial volume of those plantations was greater than that estimated by the business plan for Mexican cedar on plantations established in wet forest zones.

In conclusion, landowners preferred tree forest plantations used for commercial purposes. Most landowners selected Mexican cedar, a native with relatively fast growth, which is facilitated by watering, low densities, and the growth of secondary vegetation, as seen in Guaje Mocho. We recommend technical support for improved management as well as thinning; thus, the wood trade could yield more products and benefits. It is important to emphasize that producers in this region have been farmers for many years and that forestry is an activity that requires up-to-date technical and complex knowledge in order for it to continue developing.

Box 5.4 Terraces to reforest degraded lands in Oaxaca, Mexico

E. Cruz-Cruz

In Mexico, dry farming began around 5000 B.C. in the Oaxaca-Puebla region (Flannery, 1983). Pressure from the growing population led to vast forested areas, from hills and slopes, to be cleared for farming purposes, accelerating the rate of erosion. This was a major phenomenon until A.D. 1000–1530. In the Mixteca region in Oaxaca state, farmers developed the *lama bordo* system for growing annual crops, which fell into disuse after the Spanish conquest (Flannery, 1983). Since that period, the rate of erosion has been 10 mm $year^{-1}$ (Kirkby, 1973). Soil erosion and degradation were accelerated owing to changes in the agricultural systems, abandonment of cropland (maintenance for the traditional *lama bordo* terraces was stopped), and introduction of goat and sheep herds by Spanish colonists. Populations of sheep and goats steadily increased after 1540, but the rate of increase was higher in the middle of the seventeenth century. This tendency changed because of the following factors: (a) Spain demanded more food; (b) the population of native people declined to its lowest level, and croplands were abandoned while shrub communities grew; and (c) the Spaniards were more interested in raising goats and sheep on those abandoned lands (Romero, 1990).

Disturbances such as deforestation, farming, and grazing, changed plant communities, ranging from a matrix of wide completely denuded areas to occasional small pristine patches. Some of the latter plant communities still exist in remote areas of the Mixteca (**Fig. 1**). Currently, rangelands in Oaxaca are overgrazed due to continuous and heavy grazing, intensive herd management (everyday household-rangeland-household) and a high stocking rate, which is about three times higher than the rangeland's capacity. Such extreme overgrazing constitutes one of the most important worldwide examples of land degradation that has become a permanent and almost irreversible process. Because of social and economic factors the problem is highly complex.

Box 5.4 (cont.)

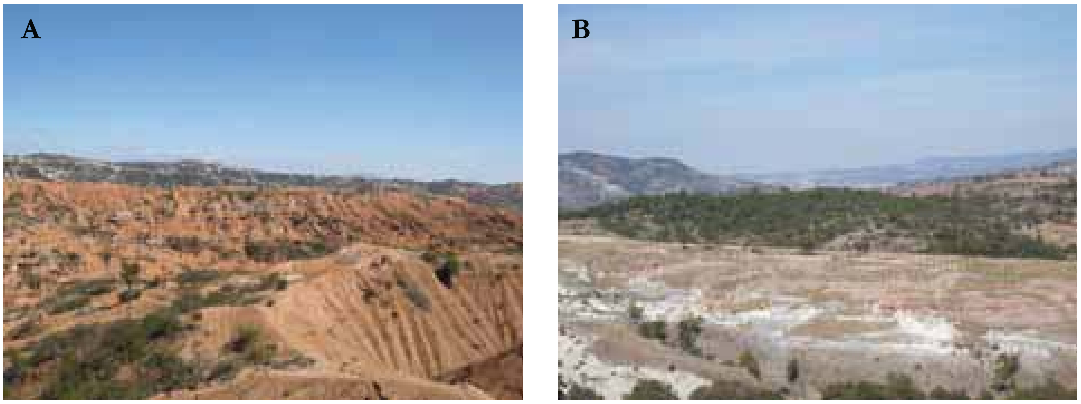

Figure 1 Soil erosion in the Mixteca region in Oaxaca State, Mexico: (A) Deep gullies with steep slopes with dispersed trees of the original plant communities in the background; (B) Cemented layer caused by carbonate accumulation supporting a pine plantation on the right. Photos: E. Cruz-Cruz

The land degradation gradient commonly looks like a pattern of successive concentric circles. Soil erosion is severe near the villages and reduces gradually farther way. Local inhabitants believe that some native plants are disappearing in those areas because of goat grazing. However, other factors are likely to be more important in driving this process, such as: high animal population, mixed herds (improved forage utilization), sedentary pastoral system, age of herders (children and elders travel shorter grazing routes), non-favourable physiographic characteristics of the grazing areas and continuous extraction of firewood.

Several soil factors including low moisture retention capacity, poor content of nutrients and organic matter, common presence of a cemented top layer and low aggregate stability, together with a scarce and low precipitation regime, an extended dry season (7 months) and the steep slopes, constrict the success of reforestation. Soils have lost their potential for production and it is probable that a state change has occurred. A threshold has been crossed in these areas. Thus, the integrity of the ecological processes has changed, and the natural stands have different potential sets of plant communities (Strigham *et al*., 2003). The upper and most fertile soil layers have been removed and the infertile and cemented C horizon is exposed. On the hillsides, the gullies dominate the landscape and the steep conditions reduce the possibilities for plant growing and establishment (**Fig. 1**).

Reforesting degraded lands

Restoration programmes in the Mixteca region were started by the *Comisión del Balsas* in 1947 and the *Comisión del Papaloapan* in 1960 (Ruiz, 1996), and were based on watershed management. Both agencies applied widespread conservation practices, these included compressed reforestation (mainly with exotic species), terracing (bank type, narrow base), exclusion of domestic animals (cattle, goat, sheep and horse), and social organization. Success was very limited because reforestation was not adequately planned to reduce soil erosion, the planted species were exotic and ecologically not adapted, and the plantation method was inappropriate and depleted plant survival.

Peasants in the Mixteca region are used to planting forest trees and they do so to create 'green zones', but without a particular conservation aim (i.e. rather their objectives are erosion control, landscape improvement, springs protection and recharge, forage production). In order to delineate a collective goal, local people must define: (a) the purpose of reforesting; (b) what

Box 5.4 (cont.)

species to choose from those appropriate according to the objectives; (b) choice of plantation method to ensure plant survival and growth; (c) social and financial organization for watching and treating disturbance agents; and (d) the future use of the plantation. There is also a need to diversify species used and to favour native ones, which have more probability of success in the local harsh and marginal conditions than exotic species.

In order to assure a high percentage of plant survival when reforesting degraded soils, it is necessary to increase moisture availability and soil retention. In Oaxaca, two plantation systems (trench-bank, common hole) resulted in an increase in moisture around the root system. If reforestation with terraces is selected, expensive costs must be prevented and there is a need for ease of construction and low labour requirements for maintenance. Terraces are useful to control soil erosion, increase infiltration and reduce the slope length. The runoff can be intercepted and kept at the bottom of the channel (**Fig. 2**). According to field evaluations, the levelled curve terraces increase the moisture content in the soil between 3 to 4%.

Terraces are built following the contour curves and the distance among them must be determined on the basis of the land form and slope. To facilitate the construction of terraces, users can use a deep plough with an agricultural tractor following a line, which was traced previously. The furrow depth should be of 50 to 60 cm. Plants are to be planted at the slope of the bank. The distance between plants depends on the plant life form: 2 m for shrubs and 4 m for trees. Both groups of species may be mixed by including a shrub between two individual trees.

Figure 2 Terraces for reforesting degraded soils: (a) here the terrace is visible as the channel at the centre of the photograph, which is the area for rain water capture; (b) plants established on a terrace; rainwater collects on the right hand side of the terrace. Photos: E. Cruz-Cruz

Box 5.5 Replacement of a forest stand of exotic species by native plants in the dryland landscape in central Chile

C. Echeverria and G. Bolados

The Chiletabacos factory, a British American Tobacco operator, is located within the Casablanca valley, an area increasingly famed for its vineyards and white wine production in the coastal zone in central Chile. As a result of urban and agriculture expansion there is very little dryland forest remaining in the valley. Chiletabacos (CT) owns 70 ha of land that was planted with *Eucalyptus globulus* about 20 years ago. The trees have no commercial value but serve as a useful buffer that screens the factory from the local community. This community uses the Eucalypt plantation as an area for grazing cattle and horses. At present, CT relies on three wells situated within the plantation for its water consumption. The company has faced serious water shortfalls in recent years and future problems with the existing wells could prove to be very costly. The restoration to native forest will better secure this ecosystem service for the future as well as reducing the risk of fire. On the other hand, the replacement of *Eucalyptus* by native species will favour biodiversity conservation and will enhance forest connectivity in this highly fragmented forest landscape.

A long-term restoration experiment of 1.6 ha was initiated in 2006 to investigate the site conditions required to restore the eucalypt plantations to native species in a dryland landscape. Within the *Eucalyptus* plantation, three different site conditions exist: (i) *Acacia caven* shrubland (*Acacia* site), (ii) open site dominated only by herb species (open site), and (iii) site cover >50% by *Eucalyptus* trees (Eucalypt site). On each site, we established six plots of 10 seedlings for each species selected: *Quillaja saponaria, Colliguaja odorifera, Maytenus boaria* and *Baccharis linearis*. On the *Acacia* site, 20 *Eucalyptus* trees that were growing between the *Acacia* trees were cut down and transported out of the area. Similarly, some trees were cut down in the interior of the *Eucalyptus* stand in order to create gaps for the establishment of the native plants. Plants were established in July 2006. Following the planting, a monitoring programme was established to measure survival every month. Also, air temperature, soil temperature and relative humidity were measured every two weeks to determinate the interespecific variation and relate the survival of the species to the climate of the area. The survival of the species was analyzed by fitting survival curves for each site condition. This was conducted using the Survival module of the statistical package R project and the Kaplan Meir distribution. The survival curves were fitted considering the species as the explanatory variable and the month of death as the response variable. Significant differences in survival were analyzed using a statistical test at 95% of significance with census data (living plants were recorded at the last sampling). A total of 32 months of survival monitoring until February 2009 was analyzed in this study.

The *Acacia* site presented the highest rate of survival with 52%, followed by the Eucalypt site with 47%. The survival rate at the open site reached up to 17% (**Fig. 1**). All of the sites exhibited a decline in the survival curves during the first summer, which was characterized by a long dry season (between months 5 and 8 in **Fig. 1**). In particular, the survival at the open site decreased from 90% to 57% (**Fig. 1**). From month 11 onwards, survival was almost constant until the last measure. In the case of *Acacia* and Eucalypt sites, the mortality rates were of 13% and 18% respectively during the whole study period.

At the species level, *B. linearis* and *Q. saponaria* registered the highest survival rates on the *Acacia* site, while on the Eucalypt site *C. odorifera* was the species that exhibited the highest survival along with *B. linearis* (**Table 1**). On the other hand, *M. boaria* presented the lowest percentages of survival for all the sites (**Table 1**), especially on the open site where any plant survived. The high rate of mortality on the open site was associated with the highest maximum temperature and solar radiation. In the Eucalypt site, the shade favoured the survival of the plants by decreasing the maximum temperature; however, it is probable that the allelopathy of the Eucalypts may have had negative effects on plant survival. On the *Acacia* site, the low level of mortality for all the species might be related to the positive effects of nurse plants provided by the *A. caven* that provides shade, humidity and nutrients to the seedlings.

Box 5.5 (cont.)

Our results reveal the potential for restoration of *Acacia caven* shrubland, which covers a large area in central Chile. Most of this vegetation is severely degraded and restoration efforts should be undertaken in the near future. *B. linearis* and *Q. saponaria* are key species that may play an important role in recovering ecosystem functions. In case of restoring deforested sites in drylands, it is necessary to adopt some silvicultural techniques (both in the nursery and in the field) to reduce the mortality of plants in summer.

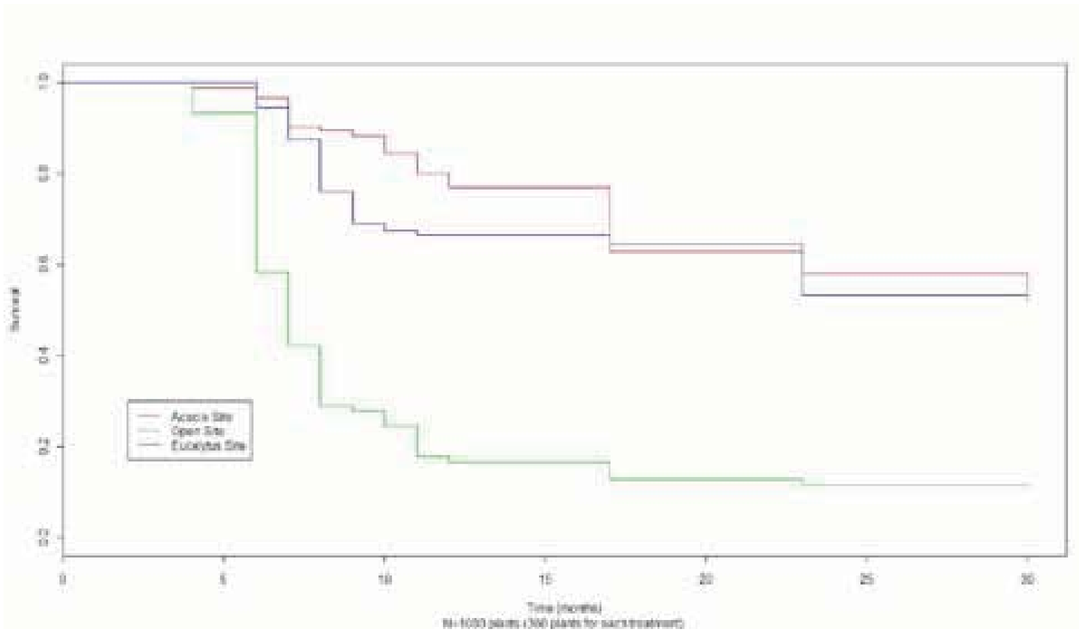

Figure 1 Survivorship curves for the restoration trials in the *Eucalyptus* plantation in central Chile.

Table 1 Percentage of survival of each species on each site in the Casablanca valley, central Chile.

Species	***Acacia* site**	*Open site*	***Eucalyptus* site**
Baccharis linearis	66.7	46.7	57.5
Colliguaja adorifera	46.7	3.3	40.0
Maytebus boaria	36.7	0.0	12.5
Quillaja saponaria	73.3	10.0	37.5

Box 5.6 Early secondary succession as passive restoration in initial stages of ecological restoration of tropical dry forest

G. Williams-Linera and E. Ascension Hernández

Although the processes of tropical dry forest recovery (TDF) are still largely unknown, there has been a recent growth in interest in secondary succession for ecological restoration. Secondary succession research would provide information on key ecological procedures and species able to enhance forest recovery. Passive restoration of TDF, i.e. the regeneration of natural forest after agricultural land is abandoned, may be used in an initial stage before active restoration takes place. Whether from an ecological or a conservation perspective, assessing the recovery of TDF needs to include an evaluation of changes in functional groups of species according to their successional status and seed dispersal modes. The objective of this study was to assess vegetation structure, woody species composition, diversity, and successional status as well as the dispersal mode of the woody species at very early successional sites.

The study area is located in the tropical dry forest region of central Veracruz, Mexico. In this area, fallow periods are short, in general no longer than 7–10 years, and old successional sites are scarce or absent. We learned that productive activity resumes on fallow land after eight years, making it practically impossible to find older successional stages. We selected five early secondary successional sites between seven to 72 months after abandonment (**Table 1**). Fallow age and land-use history were determined based on information from local inhabitants. Although the sites were not continuously used or burned every year, they have gone through a series of repeated intermediate short fallow periods that were not recorded.

At each successional site, all woody vegetation was surveyed in sixteen 5 x 5 m plots adjacent to the restoration experiments described in this chapter. Total basal area and density at early successional sites was 0.40 to 3.88 m²/ha, with 900 to 5450 individuals/ha. Mean height varied from 1.0 to 2.1 m (**Table 1**). Community diversity was evaluated as richness (number of species, S), Fisher's alpha, and Shannon's diversity index (H). Forty-five woody species were recorded at the early successional sites. Richness varied from eight to 21 woody species per site (**Table 1**). The relative ecological importance of each species at each site was expressed as an importance value index (IVI) calculated by averaging the values of relative dominance, relative density, and relative frequency. Species with the highest IVI values are shown in **Fig.1**.

Tree species were classified according to the light growth conditions required by their juveniles. Successional status was primary species or shade intolerant (growing exclusively or preferentially in TDF), shade intolerant species as pioneer and secondary tree species (establishing in cleared areas and only persisting under high light conditions), and intermediate species. The 45 species were classified in those broad categories: 23 were primary species, while 13 were intermediate and occur naturally in dry forest but also grow at early and intermediate successional sites. Only six species were clearly shade intolerant or pioneers (**Table 1**). Tree species were grouped according to seed dispersal syndrome as either zoochorous (seeds dispersed by animals), anemochorous (seeds dispersed by wind), or autochorous (seeds self-dispersed by gravity and ballistics). Woody plants were predominantly zoochorous (31 animal-dispersed species), followed by autochorous (six self-dispersed) and anemochorous (five wind-dispersed species) (**Table 1**).

In our early successional stages, differences in vegetation structure and species dominance patterns may be attributable to time since abandonment but possibly depend on the particular land-use history and disturbance level of each site prior to abandonment. Additionally, forest recovery potential may be affected by environmental factors (distance to nearby forest and seed accessibility, water proximity, topography) or anthropogenic ones (proximity to dirt roads and human settlements or the presence of trees in the pastures and living fences) related to regional TDF heterogeneity and high beta diversity (Williams-Linera and Lorea, 2009). The few dominant species in most early successional sites were *Acacia cochliacantha*, *A. cornigera*, and *Guazuma ulmifolia* (**Fig. 1**). Dominance by the Fabaceae family and the importance of *Guazuma ulmifolia* in early successional sites has been reported for most TDF successional studies in Central America.

Box 5.6 (cont.)

Our early successional sites had already recruited several of the intermediate and primary species, indicating that forest species were entering the successional process at very early stages. Some of the species recorded at early successional sites and also found in regional forests were *Brosimum alicastrum*, *Bursera simaruba*, *Croton reflexifolius*, *Ipomoea wolcottiana, Leucaena lanceolada*, *Maclura tinctoria, Malpighia glabra,Pisonia aculeata Randia aculeata*, *Spondias purpurea, Tabebuia chrysantha*, *Thouinidium decandrum*, and *Trichilia trifolia* (Williams-Linera and Lorea, 2009).

At early TDF successional sites, the recruitment of individuals from re-sprouting is very high and therefore a significant mode of forest regeneration. Tree species that sprouted at our successional sites were *Caesalpinia cacalaco*, *Diphysa carthagenensis, Guazuma ulmifolia*, *Ipomoea wolcottiana*, *Leucaena lanceolata*, *Maclura tinctoria, Malpighia glabra*, *Pisonia aculeata, Tabebuia chrysantha*, *Thouinidium decandrum*, and *Xylosma velutina*. Interestingly, these tree species have the potential to colonize disturbed areas and can therefore be considered potentially important in ecological restoration trials in the region. The entry of mature forest species into the successional process at very early stages and the recruitment of individuals from sprouting may facilitate the recovery of the dry forest in Veracruz. Overall, there is evidence to suggest that early successional stages can lead to the recovery of TDF in central Veracruz.

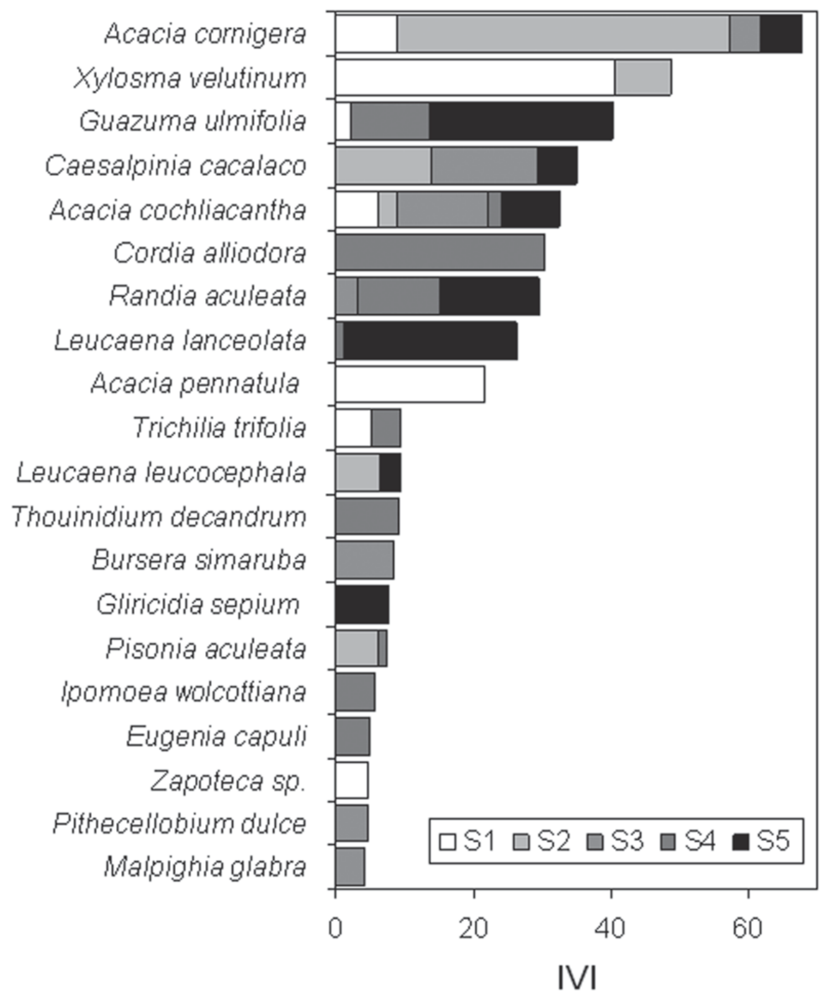

Figure 1 The 20 most important tree species recorded in five very early successional sites in the tropical dry forest region of central Veracruz, Mexico.

Box 5.6 (cont.)

Table 1 Characteristics of five very early successional sites in the tropical dry forest region of central Veracruz, Mexico. Variables are fallow age (months), basal area (m^2/ha), density (individuals/ha), and height (m). Richness is number of species recorded on 16 5 x 5 m plots per site, individuals in sampled area, Fisher's alpha, Shannon diversity index, and number of species classified in a successional status or seed dispersal mode.

	S1	S2	S3	S4	S5
Site	Hato Marines	Rinconada	Don Tirso	Xocotitla	Dios Tigre
Fallow age	7	8	36	48	72
Basal area	1.97 (0.52)	0.40 (0.21)	2.46 (1.77)	3.88 (0.45)	2.74 (1.13)
Density	2225 (505)	900 (336)	4750 (859)	5450 (565)	2925 (436)
Height	2.07 (0.23)	1.8 (0.16)	0.96 (0.27)	1.79 (0.06)	1.80 (0.31)
Richness	11	8	19	21	10
Individuals	87	36	189	218	116
Fisher's alpha	3.33	3.19	5.27	5.73	2.62
Shannon Index	1.88	1.37	2.28	2.05	1.63
Successional status					
Primary	3	3	7	10	5
Intermediate	2	2	6	4	2
Pioneer	5	2	2	5	3
Seed dispersal mode					
Animal	7	4	13	12	6
Wind	1	0	0	5	0
Self	2	3	2	2	4

Box 5.7 Tropical dry forest restoration in Chiapas, Mexico, and basic knowledge for native tree species: phenology, seed germination and seedling growth

Á. G. Becerra Vázquez, N. Ramírez-Marcial, S. C. Holz

Forest restoration is an option that must be reconciled with other social demands, such as production systems for home consumption and the market and the allocation of extensive areas for conservation (González-Espinosa *et al*., 2007). One of its strategies is the acceleration of natural forest regeneration in tandem with forest resources to provide economic and social value (Parrota *et al*., 1997). This has resulted in the identified need to develop strategies and methodologies for sustainable management of forests through the use of native species of economic value for the recovery of functionality and productivity of forests. The scarce information about initial stages in the biology of native tree species is one of the limitations for dry forest recovery. Some required studies deal with phenological stages, seed germination and seedling growth for selected species that are useful for local people, mainly for fuelwood and poles (see Chapter 6). We evaluated these variables in nine deciduous tree species from forest remnants located in the Central Depression of Chiapas, Mexico. The study site is located at the buffer zone of the El Ocote Biosphere Reserve (16° 53' 52" and 16° 50' 47" N and 93° 27' 28" and 93° 24'17" W). The climate is warm to sub-humid with a mean annual precipitation of 1100 mm with summer rains. The dry season extends from November to May. Average annual temperature is 22°C. The elevation ranges between 820 and 980 m. Soils are rendzina and fine textured lithosols. Populations are mainly from the Zoque indigenous group. The main economic activities include traditional agriculture, with maize, beans, pineapple, banana, and coffee, as well as cattle ranching. There are also other activities that demand large quantities of firewood, such as pottery, cooking, and the toasting of coffee.

Flowering of most species in TDF occurred between January–May. In general, fruiting in most studied species was observed during several months, particularly of *Acacia pennatula*, *Bursera simaruba*, *Magnolia mexicana*, *Ternstroemia tepezapote*, and *Trichospermum mexicanum*. The ripening of fruits and seed dispersal was higher from the end of the dry season. Seed germination and seedling survival were significantly different among studied species ($p < 0.05$). *Acacia* and *Erythrina* started germination earlier and completed 95% of germination in ca. 35 days, whereas other species delayed germination between 30–70 days and presented higher variation in germination percentages among species. Final seed germination (SG) was greater than 40% for seven out of nine species (except *Trichospermum mexicanum* and *Leucaena leucocephala*, which had SG below 24%), which is considered adequate for purposes of propagation in nurseries, but the high variation in SG denoting the speed and synchrony in germination (Mean Time of Germination, MTG and Germination Value GV) were deployed in *Acacia pennatula* and *Erythrina goldmanii* (MTG = 11.5 and 13.3 days, and GV = 34.7 and 22.7, respectively), so this could suggest the presence of some mechanism of seed dormancy in other species (MTG which presented over 57 days and VG under 1).

Seedling survival was relatively high (90–100%) for most species (except for *T. mexicanum*, 76%). Maximum height and stem diameter at ground level were measured for 50 seedlings obtained previously for germinated seeds. Marked differences in RGR were found among species (**Fig. 1**). The highest relative growth rates, RGR were presented by the pioneer *Trichospermum mexicanum* that displayed twice the growth of late-successional species (such as *Magnolia mexicana* and *Sideroxylum* sp). These results suggested that all studied species are easy to propagate and therefore have a high potential to be used for purposes of forest restoration. For the implementation of specific restoration projects it is necessary to consider the usefulness of the species assigned by the local human population and above all it is necessary to incorporate these and other species in different forest management models based on the biological characteristics of each one of them.

Box 5.7 (cont.)

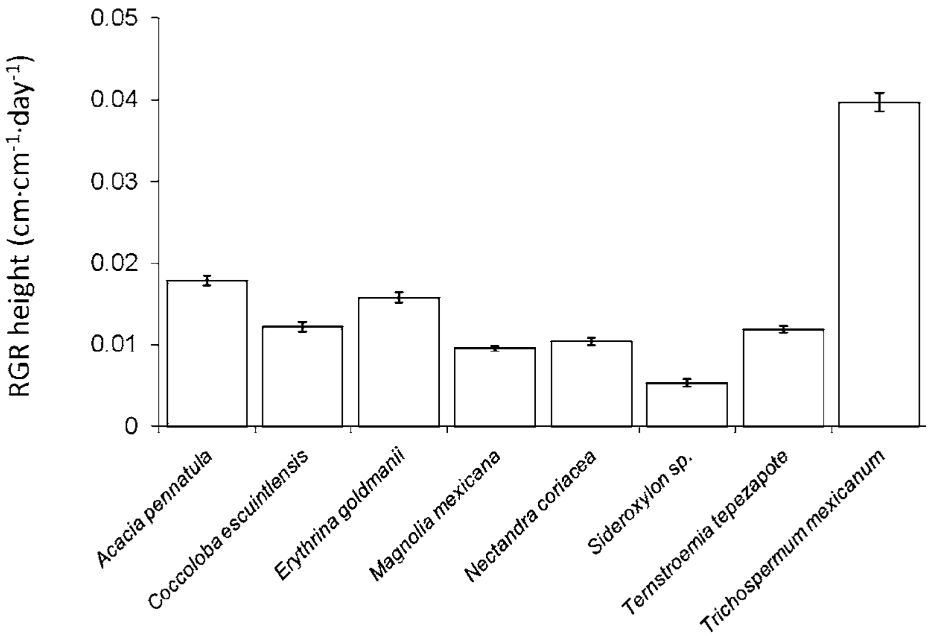

Figure 1 Relative Growth Rates in height (RGR, cm/cm/day) for eight tropical dry forests species evaluated under partial shade in Ocuilapa de Júarez, Chiapas, México.

Box 5.8 Soil seed bank, seed removal, and germination in early secondary succession of a tropical dry forest region in central Veracruz

L.P. Barradas-Sánchez, O.O. Ponce-González, C. Alvarez-Aquino

Artificial establishment of tree species and encouragement of natural regeneration are the main practical approaches to tropical forest restoration. Artificial establishment requires a considerable financial investment, whereas natural regeneration or passive restoration might be a slow but relatively inexpensive method that represents an option when large areas previously used for grazing or agriculture have been abandoned. However, a successful approach depends on an understanding of forest regeneration capacity and the key process that facilitates or inhibits it. Seed density and soil composition in the seed bank, predation, and germination capacity are important aspects of forest recovery in abandoned areas.

Seed density and floristic composition were evaluated in the soil seed bank in order to assess the potential for vegetation recovery. Ten soil samples (30 cm x 30 cm and 5 cm depth) were randomly collected at five abandoned sites (previously occupied by cattle or agriculture) and in two forest fragments. Samples were collected at the end of the dry season and transported in black plastic bags to the laboratory prior to the germination assessment. Soil samples were spread over 70 plastic trays (53 × 26 cm) in a thin layer and were watered every 1–2 days. Seed density and composition were determined by observing seedling emergence twice a week over 7 months. Emerging seedlings were counted and identified in terms of genus or species. Species were identified through field observations, and comparison with herbarium specimens. A total of 3946 seeds germinated in trays, most of the seeds corresponding to grass and herbaceous

Box 5.8 (cont.)

species; woody species were scarce, with only the following species present: *Acacia* spp., *Piscidia piscipula, Bursera cinerea, Cesalpinea cacalaco, Croton reflexifolius, Gliricidia sepium*, and *Ipomoea wolcottiana*. The highest density was registered at the site previously used for agriculture (1303 seeds/m^2, mostly grasses) followed by those used for cattle (701, 707, 704, and 411 seeds/m^2), whereas the lowest densities were recorded in the forest fragments (458 and 101 seeds/m^2, without woody species). Germinated seeds corresponded to 69 species, 49 genera, and 22 families. The best represented families with the highest number of species were Euphorbiaceae and Asteraceae. Cyperaceae was the family with the highest number of individuals.

Percentage and germination rate were evaluated in the field and in the laboratory. Seeds were tested with and without mechanical scarification with sandpaper. The selected species were native and common in the study area (*Acacia cochliacantha, Caesalpinia cacalaco, Ipomoea wolcottiana*, and *Senna atomaria).* In field experiments, they became established in tree environments typical of tropical dry forest in Veracruz: pasture, secondary vegetation, and forest fragment. Seeds were protected from predation with metallic netting. In the laboratory, germination was performed in controlled light and temperature conditions similar to those of the field (25–35ºC and 12 hours/day). Seed removal was evaluated in the field under the mother tree. Tested species were the same used for germination evaluation. Exclusions made from metallic netting of different sizes were used to test seed removal; treatments included total access, rodent exclusion, and insect exclusion. Germination percentage, both in the field and the laboratory, was higher when seeds were mechanically scarified. In the field, *Ipomoea*, *Cesalpinia* and *Acacia* yielded a percentage of ca. 50% (97, 57, and 47%, respectively). The exception was *Senna*, with a low value (10%). There were no differences among environments. Under controlled conditions, germination percentages were elevated (*Ipomoea* 99%, *Cesalpinia* 98%, *Acacia* 81%, and *Senna* 18%). In general, seed removal was below 30%, the exception being for *Senna*, with over 50%. As expected, the highest value was recorded for total access and the lowest for insect exclusion. However, any of the values represent a significant seed loss.

Results suggest that the contribution of the seed bank to natural regeneration is not significant, at least in early succession. Seed removal was not a constraint and woody species might germinate in the field, although the scattered periods of rain complicate early seedling establishment. On the basis of germination percentage in the field and in the laboratory as well as the seed removal rate, all of the selected species can be used in restoration plans in the area. Seeds can be germinated *ex situ* and grown for some months in nurseries prior to seedling transplant – or they can be planted directly during the rainy season. Of the selected species, *Ipomoea* appears to be particularly promising for restoration plans because it is present in the seed bank, exhibits high *in situ* and *ex situ* germination, and has a low removal percentage. Field observations during the dry season suggest that seeds of this species are removed by ants near nests where they can germinate in high numbers during the first summer rains. In contrast, *Senna* is recommended for germination in controlled conditions and presents a low potential for direct seeding because of the high seed removal rate.

Box 5.9 Effects of avian ingestion on seed germination in central Chile

S. Reid and J.J. Armesto

A study of the effects of avian ingestion on seed germination of Mediterranean species in central Chile provides evidence of the positive effects of birds on seed germination with prospects to facilitate the regeneration of sub-Andean shrublands. Given that a mean of 14 plant species, i.e. 34.3% (range 10.5 to 53.1%) of the total woody flora in a gradient from dry to wet sites in central Chile bear fleshy fruits dispersed primarily by birds (Hoffmann and Armesto, 1995), we evaluated the effect of avian ingestion on seed germination of five woody species whose seeds are commonly consumed by a few species of birds in central Chile. We compared the responses of bird-defecated seeds to manually-extracted seeds and seeds surrounded by intact pulp in controlled laboratory conditions.

Box 5.9 (cont.)

Collection of seeds for germination assays was conducted in the Estación de Investigaciones Ecológicas Mediterráneas (EDIEM) which lies in the Andean foothills, 20 km east of Santiago, between 1050 and 1915 m surrounded by evergreen sclerophyllous woodland. Seeds were collected from five avian-dispersed woody species, *Azara dentata* (Flacourtiaceae), *Schinus polygamus*, *S. molle* (Anacardiaceae), *Cestrum parqui* (Solanaceae), and *Maytenus boaria* (Celastraceae). These shrub species conform to 90% of the plant species found in bird droppings and are frequently found on the outskirts of the large city of Santiago (Reid, 2008). During the summer of 2006 (January to March), we collected bird droppings containing shrub seeds in the matorral of the EDIEM. Fresh fruits were also collected from a minimum of five individuals of each of the five shrub species found in bird droppings. Fruit pulp was manually removed from a number of seeds corresponding to the number of seeds extracted from bird droppings. Germination assays were conducted in glasshouse conditions set for a spring ratio of 14 hours light to 10 hours darkness. Light intensities were 350 to 500 μmol m^{-1} s^{-2} from a metal halide light measured at the outer surface of containers. Temperatures ranged from 16°C to 28°C. For each species treatments were: seeds collected from bird droppings (ingested seeds), seeds manually removed from the pulp (extracted seeds), and seeds sowed with the pulp (intact fruits). Seeds were placed on filter paper in Petri dishes, 9 cm diameter, and watered every 2–3 days with distilled water. Germination was monitored every three days by recording the emergence of the radicle during three months in the austral winter and spring (July to October 2006). A total of 774 seeds were tested. We assessed germinability (the final percentage of sown seeds germinated after three months) and germination rate defined as the number of seeds germinated per time interval.

Avian ingestion significantly increased seed germinability (ANOVA $F_{2,45} = 12.1$, $p < 0.001$; **Fig. 1**). Fifty percent of the seeds that were avian ingested germinated, while 30.5% germinated in the extracted seed treatment and 12% of the seeds germinated within intact fruits. Species-specific level analyses show that for *A. dentata* germinability did not differ between ingested and extracted seeds (Mann-Whitney *U*-test: Z = 0.09, $p = 0.93$) and no seeds germinated within intact fruits. For *S. molle*, germinability was significantly higher for avian ingested seeds than for seeds within intact fruits (ANOVA $F_{2,9} = 4.94$, $p = 0.04$; Tukey's test, $p = 0.03$; **Fig. 2**). For *C. parqui*, germinability was significantly higher for avian ingested seeds than for manually-extracted seeds (ANOVA $F_{2,3} = 10.25$, $p = 0.05$; Tukey's test, $p = 0.04$). In *M. boaria*, germinability was equal for ingested and extracted seeds (85% germinability) and half the seeds sowed with the pulp germinated. In contrast, only one out of 100 seeds of *S. polygamus* germinated in each of the ingested and the intact fruit treatments, and no extracted seeds germinated.

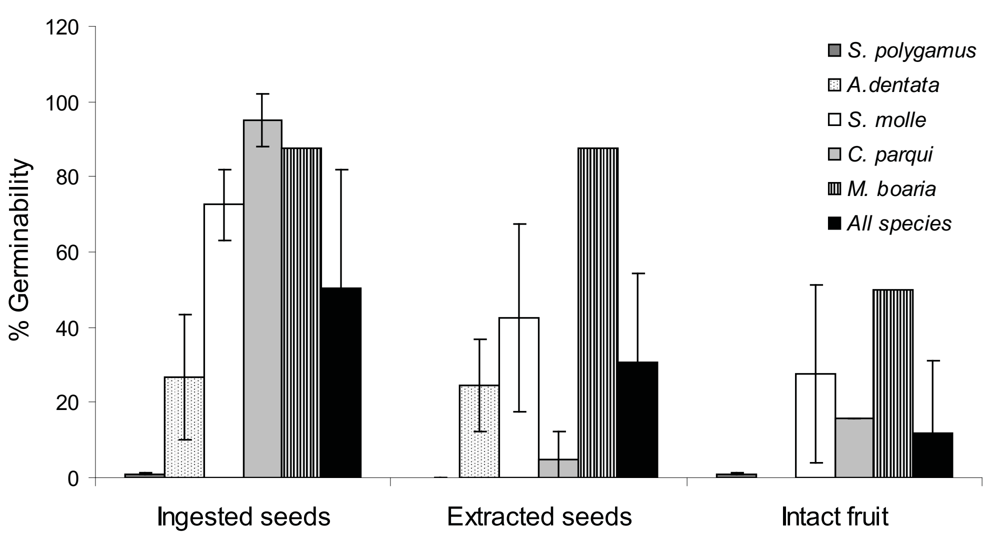

Figure 1 Final percentage of germinability for each seed treatment.

Box 5.9 (cont.)

Avian ingestion accelerated the rate of seed germination. Species-specific analysis showed similar rates for ingested and extracted *A. dentata* seeds (RMANOVA $F_{1,16} = 1.22$, $p = 0.28$ and HS, $p = 0.49$). For *S. molle*, a significant increase in germination rate was observed for avian ingested seeds compared to those within intact fruits (RMANOVA $F_{1,6} = 18.91$, $p = 0.01$ and HS, $p = 0.01$; **Fig. 2**). In *C. parqui,* the germination rate of ingested seeds was significantly faster than extracted seeds and those within intact fruits (RMANOVA $F_{1,2} = 110.45$, $p = 0.01$ and HS, $p = 0.01$; and $F_{1,2} = 108.97$, $p = 0.01$ and HS, $p = 0.02$, respectively). For *M. boaria*, avian ingested seeds had a significantly faster germination rate than seeds within intact fruits (GB = 8.88, d.f. = 2, $p = 0.01$).

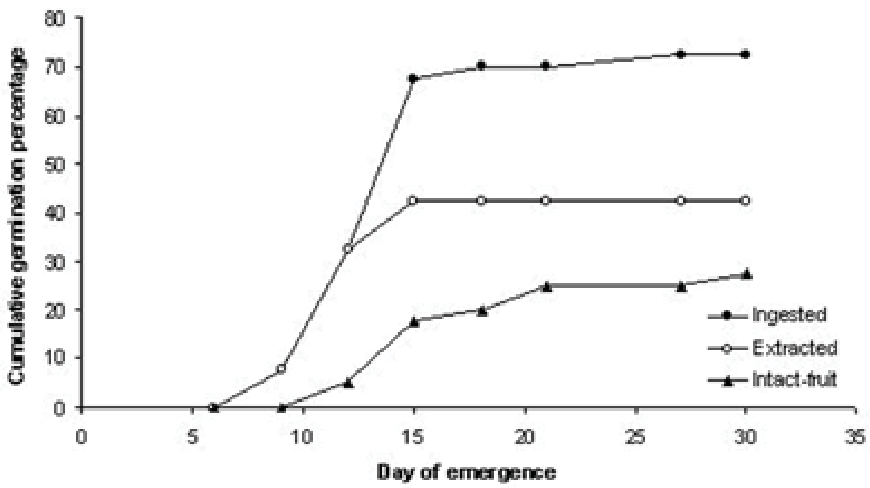

Figure 2 Cumulative seed germination percentage for each seed treatment with 40 *Schinus molle* seeds.

We concluded that avian ingestion increased seed germinability in four of the five shrub species that are commonly consumed by birds in this landscape. Although higher germinability is positively related to plant fitness (Traveset *et al.*, 2007), the dependency of plant species on avian ingestion for the completion of their reproductive cycle was variable and strongest for *A. dentata.* Given that seeds within intact fruits of this species did not germinate at all, we suggest a dependency of this species on avian frugivory to regenerate in vegetated or open areas. Alternatively, the lack of seed germination in *S. polygamus* may have been due to low seed viability (A. Sandoval, unpubl. results).

In contrast, the advantages of faster germination for seeds are less well understood. Izhaki and Safriel (1990) suggest avian frugivores add temporal heterogeneity to seed germination responses, by enhancing germination rates and shortening seed dormancy. The advantage of increased temporal germination heterogeneity in variable environments is spreading the risk of seed and seedling mortality over time. Traveset *et al.* (2001) suggest that faster seed germination allows dispersed seeds to have a shorter exposure to seed predators. Paulsen and Högstedt (2002) suggest that early emerging seedlings in spring have an advantage because they develop deeper root systems making them more resistant to drought.

This study shows that avian frugivores in this Mediterranean ecosystem do not harm seeds after ingestion and tend to speed up germination.

Box 5.10 Tree-seedling establishment in fragmented Mediterranean forests of central Chile

M. Holmgren, J.L. Celis-Diez, J.J. Armesto

Mediterranean-type ecosystems are global biodiversity hotspots on all continents. Vegetation cover is extremely fragmented by the combination of natural and man-induced disturbances. Regeneration of these semi-arid ecosystems has proved difficult but can be strongly facilitated by plant-nurse interactions. We studied whether the interaction of micro-climate and herbivores under remnant vegetation patches defines a patch-size threshold for tree seedling recruitment and whether that threshold is dependent on overall climate conditions and plant functional types.

We used a combination of correlational and experimental approaches to investigate this problem. Field observations and experiments were conducted in Andean and coastal Chilean shrublands, representing a gradient in rainfall (350 and 500 mm annual precipitation, respectively). We planted one-year old seedlings of relatively drought-tolerant (*Quillaja saponaria*) and drought intolerant (*Cryptocarya alba*) species under open and shaded conditions considering a gradient of shrub patch sizes (1, 5, 10–15, >30 m in diameter, n = 10 per patch size) and at increasing distances from the canopy edge (5 and 0.5 m outside of the border of the patch, 0.5, 2, 5, 15 m inside of the patch). Half of the seedlings were protected against mammal herbivores (mostly rabbits and horses).

We found no naturally established tree seedlings in the drier foothills of the Andes. In the moister coastal range, seedlings were frequently under the canopy of small and medium-sized shrub patches (5–15 m diameter). Seedlings in open areas were found only at the edge of large shrub patches (>30 m). Herbivore pressure by rabbits and hares is enormous in the Andean foothills where no seedlings survived in the experimental non-protected plots. At the moister coastal range site, herbivore pressure was lower and was further reduced under larger patches. Seedling mortality due to drought stress was reduced under the shrub canopy and significantly decreased with shrub patch size, particularly in the drier Andean site. Large shrub patches are cooler and moister which ameliorates plant thermal and water stress particularly at the drier site. Seedling survival was strongly linked to physiological performance under different water and irradiance conditions.

Our results indicate that shrub fragmentation might be irreversible at the drier end of semi-arid ecosystems as shown in the Andean foothills of central Chile. Conservation of large remnant shrub patches in the landscape, wherever possible, is essential here to facilitate ecological restoration by combining herbivore exclusion and shrub shade. Under moister conditions (coastal sites and wet years), herbivore protection may be sufficient to enhance tree regeneration. In such areas, seedling establishment will also increase along large patch edges and under existing small shrub patches. ENSO events in central Chile increase precipitation to levels comparable to those found in the coastal region, and could potentially increase the probability of seedling establishments in Andean foothills.

Figure 1a, b Discrete shrub patches separated by open areas of herbaceous cover have replaced continuous evergreen shrubland cover in Mediterranean Chile (Photos: M. Holmgren).

Box 5.11 Post-fire restoration of native tree species: effects of wood shaving application

M. Lallement, C. Tognetti, M.E. Gobbi

Fires are the most devastating anthropogenic disturbances in forests of the Andean-Patagonian region. They severely affect surface soil physical and chemical characteristics, as well as vegetation and fauna, thus increasing the risk of erosion. These forests are very important for the conservation of biodiversity, climate and watershed regulation, and soil stability. Therefore, it is relevant to implement strategies that favour ecological restoration after a disturbance has occurred.

Reforestation with native species is one of the most used restoration strategies in the Andean-Patagonian forests. This strategy initially depends on the availability of plants, both in terms of quantity and quality. The adequate growth and development of these plants will determine the success of a restoration project. Aspects such as the type of substrate used for seedling production and the strategies used to plant these seedlings in the field should be carefully considered when planning restoration projects. The general objective of the work carried out here was to evaluate strategies that facilitate the recovery of burned forest areas of the Nahuel Huapi National Park (northwestern Patagonia), specifically by means of reforestation with native tree species. For this, we evaluated:

i. The success of a reforestation project that included the aid of volunteers to plant three native tree species (*Austrocedrus chilensis, Nothofagus pumilio* and *Lomatia hirsuta*) in a post-fire area of the Nahuel Huapi National Park.

ii. The effect of wood shavings, either applied as mulch or incorporated into the soil, on the survival and growth of *A. chilensis, N. pumilio* and *L. hirsuta*: (i) in a nursery, and (ii) in the field.

iii The effects of wood shavings, either applied as mulch or incorporated into the soil, on the water dynamic of a burned volcanic soil.

The three tree species selected for this study are characteristic of the xeric forests in which most of the fires in the region occur. They have different life-forms and reproductive strategies (**Table 1**). Regarding *L. hirsuta*, there are no previous records of this species being used in restoration projects. The study was carried out in a post-fire shrubland in the Challhuaco Valley, where mean annual precipitation is 1000 mm (mostly rain and snow in autumn-winter). Seedlings were obtained at local nurseries, and their ages varied as follows: 3–5 years old (Objective I), 1-year old (*A. chilensis* and *L. hirsuta*, Objective II), and 3-years old (*N. pumilio*, Objective II). In the field study, seedlings were planted under shrubs (nurse plants).

Growth and survival of plants in the reforestation project aided by volunteers (**Fig. 1**) was satisfactory; values obtained were within the range of those obtained in previous studies by experienced personnel and under similar climatic conditions. The highest survival after the first year was for *A. chilensis* (51%), followed by *L. hirsuta* (43%), and *N. pumilio* (29%). In conclusion, reforestation project aided by volunteers proved to be a good strategy to recover degraded areas, and provided an opportunity to educate citizens on environmental problems.

To evaluate the effects of wood shavings on the survival and growth of the three species, pine (*Pseudotsuga menziesii*) wood shavings were either incorporated into the soil at a 1:3 shavings to soil ratio (v/v) or mulched over the soil forming a 2 cm deep layer. As controls, soil (burned for the field assay, unburned for the nursery assay) with no wood shavings was used. In the field, seedlings were only watered when they were planted; while in the nursery they were watered tree times a week. In the nursery assay, mixing wood shavings in with the soil increased survival of the three species and improved some of the growth indicators (**Fig. 2**). Apparently, this is a promising strategy to improve the nursery production of native species. However, mulching soil with wood shavings had no effects on survival or growth (**Fig. 2**). In the field assay, wood shavings had no effects on the measured parameter, regardless of whether it was mulched or was incorporated into the soil (**Fig. 2**). However, mulching did reduce and stabilize soil surface temperature.

Box 5.11 (cont.)

This is the first report of the use of *L. hirsuta* in restoration projects. The results outlined so far indicate that this species has good survival and growth rates, and should be considered in future reforestations.

For the assays on the dynamic of soil water (assays without plants) surface soil from a burned shrubland was used. The soil was either mixed with wood shavings (at three soils to shavings rates) or mulched with wood shavings (two mulch depths). In both cases, a burned soil without wood shavings was used as a control. Mixing wood shavings in the soil increased water draining speed and field capacity; this effect was more marked as the proportion of wood shavings increased. Applying a layer of wood shaving mulch on the soil delayed evaporation, although final soil moisture values were not affected. The effects of wood shavings on the dynamics of soil water could prove to be positive for plants, provided that the water remains available near the root system.

The application of new strategies aimed at increasing survival and growth of native plants, especially those that use low-cost and easily available resources, stimulates the development and implementation of restoration projects. Ultimately, it accelerates the succession process in deteriorated natural habitats, and improves recovery of degraded areas.

Table 1 Characteristics of the three native tree species used in this work.

	Austrocedrus chilensis	*Nothofagus pumilio*	*Lomathia hirsuta*
Family	Cupresaceae	Fagaceae	Proteaceae
Order	Coniferales	Fagales	Proteales
Common name	Ciprés	Lenga	Radal
Foliage	Perennial	Caduca	Perennial
Life-form	Tree	Tree	Shrub/Tree
Re-sprouting ability	No	No	Yes
Masting	No	Yes	No

Figure 1 Volunteers working in a native plantation in a post-fire area of the Nahuel Huapi National Park, Argentina (author's photograph).

Box 5.11 (cont.)

HI (%)	299	222	177	75	115	88	164	340	162
GI (%)	275	311	177	100	200	100	47	78	6

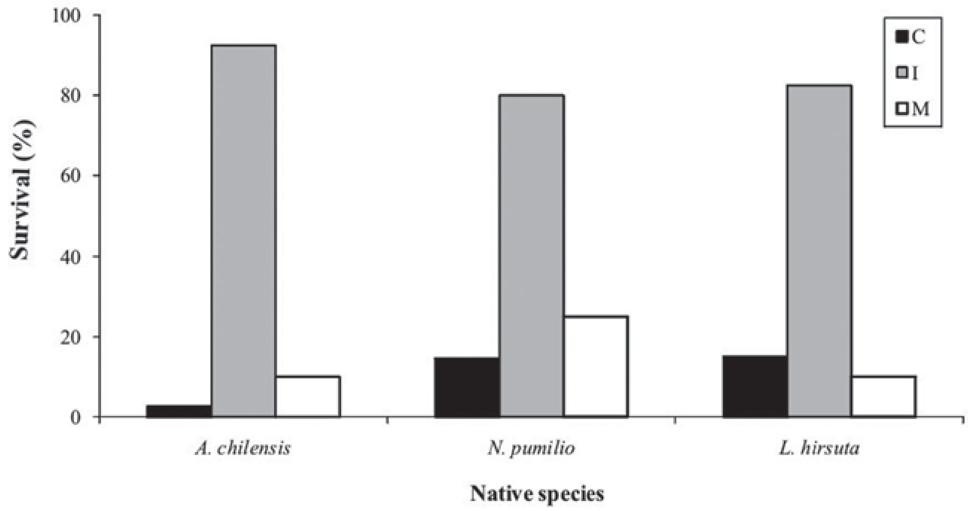

Figure 2 Relative increase of height (HI) and growth (GI), and survival at the end of the nursery assay. Growth indicators were number of whorls (*A. chilensis*), number of branches (*N. pumilio*) and number of leaves (*L. hirsuta*). C: control, I: wood shavings incorporated into the soil, M: wood shavings applied as mulch.

Case Studies

Central Valley, Chile

The study area is located in the Central Valley between the Andes and coastal mountain ranges, in the transition zone between Mediterranean sclerophyllous forest and deciduous dry forest (35°–38°S; **Fig. 5.1**). The climate is semi-arid Mediterranean. The annual average (30 year) precipitation is 330 mm during winter, with 6–7 dry months. The annual mean temperature is 15°C. Vegetation is composed of tree, shrub, and herbaceous patches that are dominated by native species (*Pasithea coerulea*, *Bromus berteroanus*, *Clarkia tenella*, *Amsinckia calycina*, *Moscharia pinnatifida* and *Helenium aromaticum*) and exotic species (*Conium maculatum*, *Centaurea melitensis*, *Fumaria capreolata*, *Carduus pycnocephalus*, *Erodium cicutarium* and *Brassica rapa*). The selected sites for field experiments are representative of different types of vegetation, soils, and human disturbance history present in the region. They were located on private lands (Los Maitenes, Las Tórtolas, and Pirque), experimental stations (Quebrada de la Plata, Calán, and San Carlos de Apoquindo) and private reserves (San Ramón watershed and Mahuida Park) (**Fig. 5.1**). The watershed has been protected from logging, cattle grazing, and woodland fires for the past 10 years by the Chilean forest service. Experiments were also conducted in the dry woodlands of Fray Jorge National Park to assess the response of species to irrigation simulating an increased frequency of wet (ENSO) years.

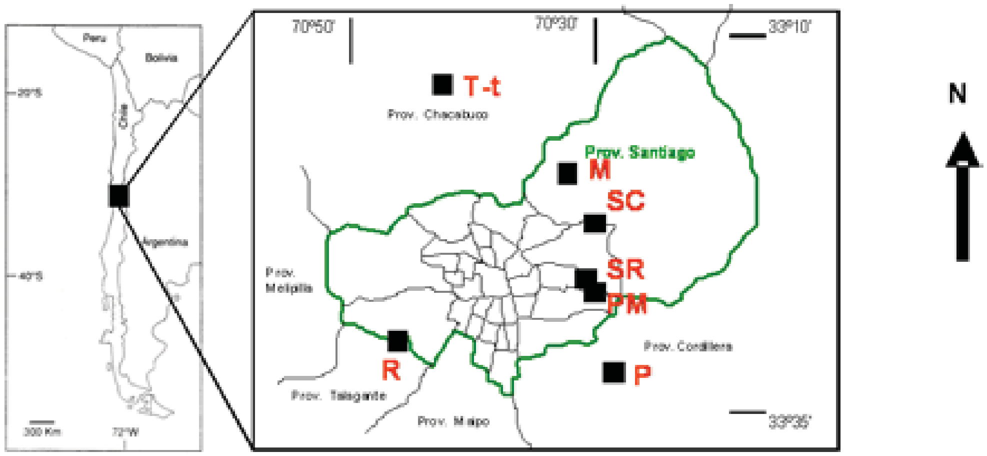

Figure 5.1 Map of the study area in the Central Valley, Chile. Restoration experiments were located in San Carlos de Apoquindo (SC), Rinconada (R), Maitenes (M), Pirque (P), San Ramón watershed (SR), Til – Til (T-t) and Mahuida Park (PM).

Experimental approaches focused on the effect of herbaceous cover, precipitation, nurse shrub species (exotic, N-fixing, and non N-fixing), herbivory (combining their effect with shrub patch cover and precipitation), the potential for restoration during rainy years, and natural recolonization of trees and birds. Experiments were established in the study area to test the effect of seven factors on woody seedling establishment of three native species (**Table 5.1**).

Table 5.1. Selected species used in the restoration field experiments in the study areas of Chile, Argentina and Mexico.

Sites	Species traits	Selected species used in restoration
Central Valley, Chile		*Colliguaya odorifera*, *Cryptocarya alba*, *Lithrea caustica*, *Quillaja saponaria Shinus polygamus*
Coastal Range, Chile		
Casablanca Valley		
	Shade tolerant	*Cryptocarya alba*, *Peumus boldus*
	Intermedia	*Lithraea caustica*, *Maytenus boaria*, *Schinus latifolius*, *Quillaja saponaria*
	Shade intolerant	*Acacia caven*, *Baccharis linearis*, *Colliguaja odorifera*
Lago Peñuelas National Reserve and Colliguay Valley		
	Shade tolerant	*Beilschmiedia miersii*, *Cryptocaria alba*, *Peumus boldus*
	Intermedia	*Maytenus boaria*, *Quillaja saponaria*, *Schinus latifolious*
	Shade intolerant	*Acacia caven*, *Senna candolleana*
Northwestern Argentina		
	Natives	*Amburana cearensis*, *Anadenanthera colubrina*, *Astronium urundeuva*,*Caesalpinia paraguariensis*, *Calycophylum multiflorum*, *Cedrela balansae*, *Coccoloba tiliacea*, *Cordia trichotoma*, *Diatenopteryx sorbifolia*, *Enterolobium contortisiliquum*, *Gleditsia amorphoides*, *Inga saltensis*, *Jacaranda mimosifolia*, *Myroxylon peruiferum*, *Pithecelobium scalare*, *Phyllostylon rhamnoides*, *Pterogyne nitens*, *Saccelium lanceolatum*, *Tabebuia impetiginosa*, *Tipuana tipu*
	Exotics	*Toona ciliata*, *Grevillea robusta*, *Eucalyptus grandis*, *E. teretricornis*, *Corymbia torelliana*, *C. citriodora*, *C. maculate*, *Flindersia xanthoxila*, *F. australis*, *Khaya senegalensis*, *Pautownia fortune*

Table 5.1. (cont.)

Sites	Species traits	Selected species used in restoration
Southwestern Argentina		*Austrocedrus chilensis*
Oaxaca, Mexico		*Acacia angustissima*, *Amelanchier denticulata*, *Cercocarpus fothergilloides*, *Desmodium orbiculare*, *Dodonaea viscosa*, *Eysenhardtia polystachya*
Central Veracruz, Mexico	Native	*Cedrela odorata*, *Ceiba aescutifolia*, *Guazuma ulmifolia*, *Ipomoea wolcottiana*, *Luhea candida*, *Tabebuia rosea*
	Selected by local people	*Chloroleucon mangense*, *Diphysa carthagenensis*, *Leucaena lanceolata*, *Lysiloma acapulcense*, *Lysiloma divaricatum*, *Maclura tinctorea*, *Cordia alliodora*, *Cedrela odorata*

The treatments were as follows:

(1) *Herbaceous cover and precipitation.* The hypothesis was that herbaceous cover in the Chilean scrubland has a negative effect on the establishment of woody seedlings, and this effect would be stronger with heavier rainfall owing to the increase in herbaceous cover and height. Results showed that the effect of herbaceous cover on woody seedling establishment differed between years. During the year corresponding to 'La Niña', an event involving extremely low precipitation, the herbaceous cover negatively affected seedling survival. In contrast, during the year with more precipitation, the effect of herbaceous cover on seedling survival was positive (**Fig. 5.2**).

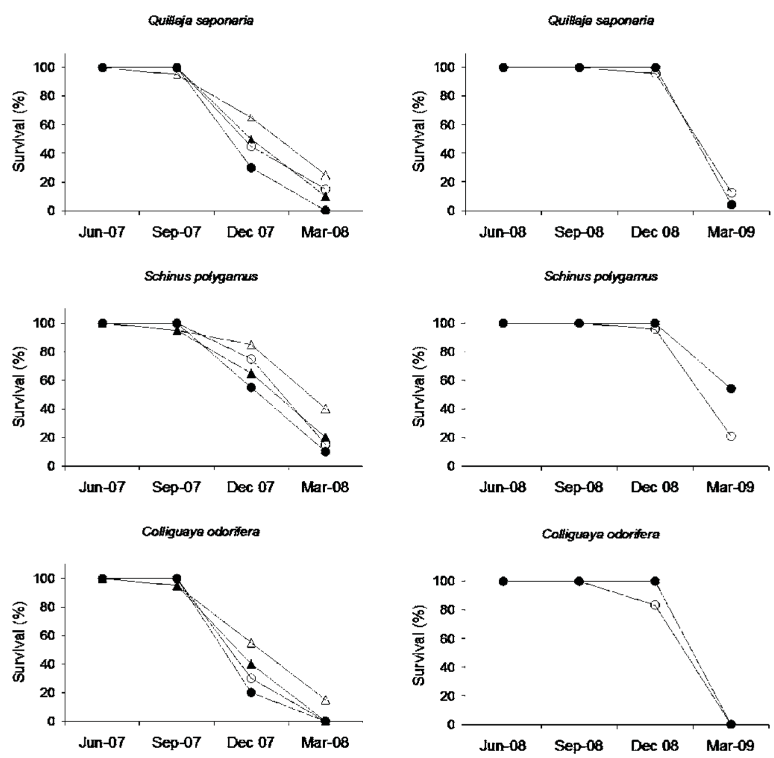

Figure 5.2 Seedling survival of the woody species planted in experimental plots in the Central Valley, Chile. Each point corresponds to percentage of seedlings alive in irrigated plots with herb cover (▲), irrigated plots without herb cover (Δ), non-irrigated plots with herb cover (●), and non-irrigated plots without herb cover (○).

(2) *N-fixing and non N-fixing nurse shrub species.* The hypothesis was that N-fixing shrub species have a stronger positive effect on the establishment of woody seedlings under their cover than non N-fixing shrub species. In the Chilean scrubland, N-fixing species correspond to Leguminosae and Rhamnaceae shrubs. Results showed that the effect was positive owing to an increase in soil nutrients; however, the effect of N-fixing species was greater than that of the non-fixing species.

(3) *Exotic tree species.* The hypothesis was that exotic trees (introduced in many areas of central Chile) have a negative effect on seedling establishment. However, the results showed a significantly positive effect; but the effect of *Eucalyptus globulus* was minor in comparison to the other exotic species.

(4) *Shrub patch cover and herbivory.* The hypothesis was that the effect of herbivory by native and exotic mammals is weaker within remnant shrub patches than in open areas and that this impact will decrease with patch size. Results showed that the effect of forest fragmentation and herbivory was negative. In smaller forest fragments and in the area nearer to fragment edge, the herbivory effect was greater.

(5) *Herbivory and precipitation.* The hypothesis was that the effect of herbivory on seedling establishment is negative, being weaker in years with higher rainfall. Results showed that the effect of herbivory and high precipitation (simulated by artificial irrigation) on seedling survival was negative.

(6) *The potential use of El Niño (ENSO) rainy years for restoration of native vegetation in central Chile.* An experiment was designed to test whether rainy years represent an opportunity for restoration. Shrub species with contrasting growth forms were subjected to a range of simulated ENSO conditions and partial herbivory by rabbits and rodents (herbivory was simulated by 'clipping'). Results showed no effect of water on survival and a strong herbivory effect. Survival was 0% and extremely low both in plots with and without herbivores. Results suggest that extending the water pulses into spring-summer, when the drought commences, will increase survival rates.

(7) *Restoration and the natural recolonization of native trees and birds in disturbed sites.* The goal was to evaluate performance of different native woody species used in forestry plantations established by private owners and to assess natural recolonization of native birds and shrubs. Results showed that natural colonization by birds and shrubs was different between planted versus non-planted adjacent sites. Within each plantation, survival and growth rates differed significantly between species, with *Quillaja saponaria* showing the highest performance. In general, the observed natural recolonization by woody species was low, whereas natural recolonization by birds was much greater. Difference in bird abundance between reforested and open adjacent sites was significantly greater in older reforestation areas, indicating that bird abundance is facilitated by reforestation.

Conclusions

According to these results, the following recommendations can be identified for restoration programmes: (1) in plantations, it is not always necessary to eliminate herbaceous cover, since only during drier years would herbs have a negative effect; (2) when plantations are established in open sites, irrigation is recommended during summer even when the precipitation of the year has been close to average; (3) plantation with successional native species should be undertaken underneath nurse species (exotic, N-fixing or non N-fixing), because they may facilitate seedling establishment. Exotic species are useful as nurses; however, in some cases (if they are aggressive) it is necessary to eradicate them after native seedlings are established. In general, nitrogen-fixing species are better nurses. (4) Despite the lower herbivory effect within fragments and on larger ones, herbivory is high enough to kill most of the seedlings, and therefore herbivore exclusion must be considered under any environmental conditions; and (5) reforestation programmes may be highly effective not only for planted species but also for other organisms such as birds, even when plantations are still relatively young.

Coastal Range, Chile

The study area is located in the Coastal Range of Valparaiso (33°S, 71°30' W; **Fig. 5.3**). Annual precipitation ranges from 100 to 800 mm. Vegetation corresponds to sclerophyllous forest, characterized by high endemism and the presence of some xerophytic plant species such as *Cactus* spp. The main land uses are agriculture, urban areas, fruit cultivation, and mining. The soil is highly degraded as a result of erosion caused by forest removal. This region is characterized by the presence of large wine exporters, leading to substantial land use changes. Also, large areas have been converted to plantations of exotic tree species. Restoration experiments on semi-arid sites were conducted in the Casablanca Valley (land owned by the Chile Tobacco Company), Lago Peñuelas National Reserve (LPNR), and the Colliguay Valley. Additionally, some plots were established in more humid micro-sites in the Colliguay Valley and the LPNR.

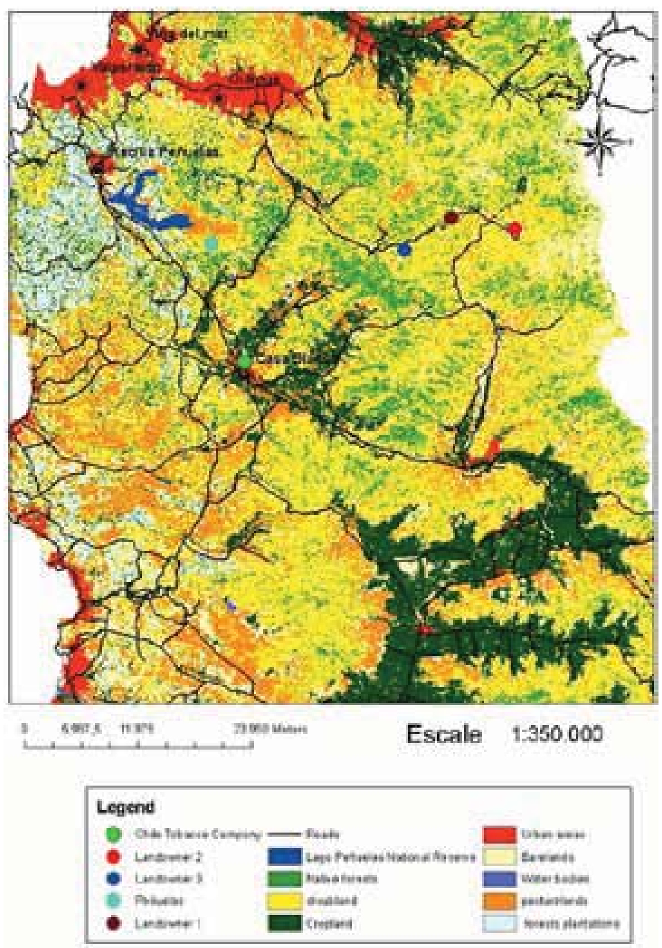

Figure 5.3 Study sites in Coastal Range, Chile. The five restoration experiments were established in Casablanca Valley, the Colliguay valley, and Lago Peñuelas National Reserve.

Field experiments were established under three site conditions: presence of *Acacia caven*, presence of *Eucalyptus* trees, and an open site. The treatments were fertilizer, *Eucalyptus* bark, and drip irrigation. The tree species established in the restoration assays included shade-tolerant, intermediate, and shade-intolerant species (**Table 5.1**). The experiments were monitored for plant health conditions every month, and growth in height and diameter was monitored every six months.

(1) *Casablanca Valley.* In Casablanca Valley, a total of 2320 seedlings have been planted since 2006. In general, the areas restored with *Acacia* and *Eucalyptus*, where soil temperature and humidity were less extreme, showed statistically higher survival and growth than the open site. The *Acacia* site presented higher rates of survival and growth for most species, suggesting that factors such as nitrogen fixation by *Acacia* could improve plant development at this site. In the presence of *Eucalyptus*, the allelopathic effects of this species could influence seedling development.

Baccharis linearis performed well under the three site conditions. *Quillaja saponaria* also had a high survival rate, especially when planted at the *Acacia* site. In contrast, *Cryptocarya alba* and *Peumus boldus* did not perform well under these conditions. Irrigation in early stages of development had a positive effect on survival and growth, especially at the *Acacia* site. With the exception of the open site, results suggested a positive effect of the combined application of irrigation, fertilizers, and/or bark on the survival of most plants. However, increased mortality of *Senna candolleana* was observed when *Eucalyptus* bark was used (**Fig. 5.4**).

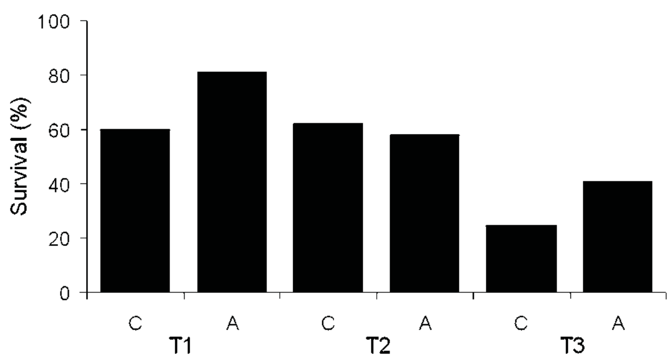

Figure 5.4 Survival percentage for the 2008 plantation between July 2008 and June 2009 for all sites (T1 *Acacia* site, T2 *Eucalyptus* site, T3 open site) in the coastal range of Chile. C is control and A is application of different treatments (bark, fertilizers and irrigation).

(2) *Colliguay Valley and Lago Peñuelas National Reserve (LPNR).* In Colliguay and LPNR in 2008, a total of 710 seedlings were established on *Acacia* sites. Additionally, four plots were established and a total of 120 individuals of species suitable for shade and humid conditions in riparian environments were planted in these areas' more humid micro-environments.

At the *Acacia* site, *Quillaja saponaria* and *Schinus latifolius* had the highest survival rates for all experiments (ca. 50 %), while *Maytenus boaria* had the lowest survival (5%). When the different study sites were compared, the highest species survival was observed at the site with conditions associated with almost no slope and the greatest influence of coastal fog. The effect of fertilizer application was uncertain; there were no clear effects on either species survival or growth rates in any experiment. However, at the more humid micro-sites, experiments without fertilizer had the lowest survival rates. *Beilschmiedia miersii* had a 40% survival rate in the Colliguay Valley, and *P. boldus* and *C. alba* around 10% in LPNR. Overall higher survival rates in Colliguay may be explained by its more humid conditions than LPNR. Results suggest that fertilizer application on humid sites increases survival; however, there is no way to ensure that seedling performance improves as a result. In Colliguay, the highest growth rates at the *Acacia* site were recorded for *Q. saponaria, S. latifolius*, and *M. boaria.*

Conclusions

In Casablanca Valley, the best species performance was recorded at the *Acacia* sites. The establishment of *A. caven, B. linearis*, and *S. candolleana* facilitated the establishment of other species on open sites while irrigation facilitated higher native plant cover at *Eucalyptus* sites. However, a conclusive evaluation of fertilizer and bark experiments would require more than a year of analysis.

In Colliguay and LPNR, the results suggested that under *Acacia* shade, the species that establish have high survival and growth rates, such as *Q. saponaria* and *S. latifolius.* At open sites, *A. caven* displayed a higher survival rate than *S. candolleana*, although the growth rate of the latter was higher. One recommendation is to use irrigation in cases where dry conditions are more restrictive. Fertilization experiment results do not indicate whether plant survival and growth were improved or not. *B. miersii, C. alba,* and *P. boldus* are recommended for shaded and more humid conditions.

Northwestern Argentina

Restoration activities focused on subtropical seasonally dry forests (SSDF) located in Salta (San Martin and Oran Departments) and Jujuy (Santa Barbara and Ledesma Departments) provinces in northwestern Argentina (22°–24°S, 63°–65°W; 350 and 750 m a.s.l.; **Fig. 5.5**). These forests include Andean premontane forest and a transition to dry Chaco forest. Chaco forest is the largest unit of tropical dry forest in the neotropics, extending through Argentina, Bolivia, and Paraguay. SSDF are largely threatened by deforestation and transformation to agriculture, mainly for sugarcane and soybean production. Timber activities are also economically important in the area and are largely restricted to selective logging of about a dozen valuable native species. The study area covers approximately 10,000 km^2 and combines highly profitable agricultural activities with local and indigenous communities living in extreme poverty. The study area harbours the highest concentrations of ethnic groups (nine) of Argentina, including groups of Andean, Amazonian, and Chaco origin. Forest transformation for agriculture is disrupting the historical forest continuity between Andean premontane forest and dry Chaco forest, creating an agricultural gap 5 to 25 km wide. In the study area, SSDF covers approximately 7,500 km^2, most of which is highly disturbed and susceptible to transformation.

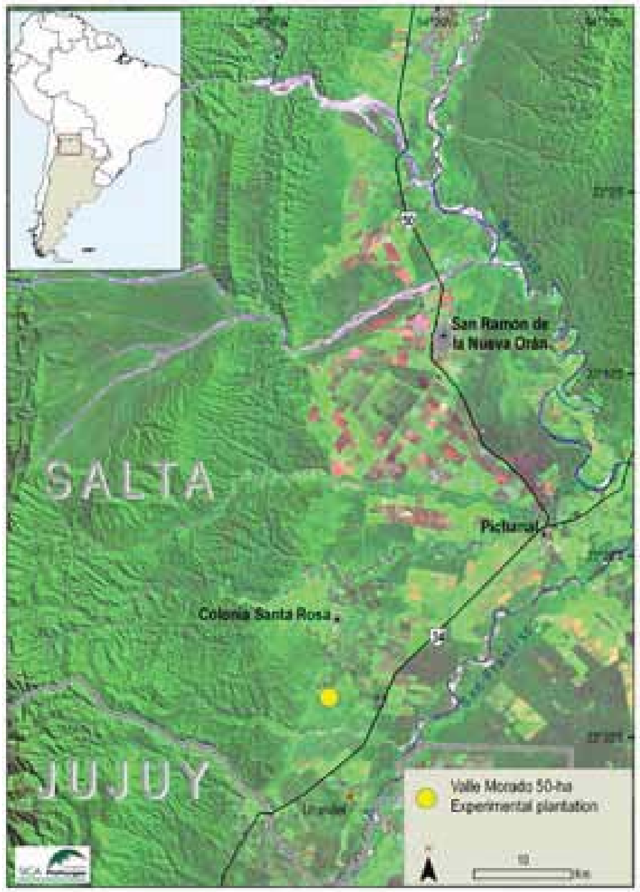

Figure 5.5 Study sites in northwestern Argentina. The 50 ha experimental plantation was located in Valle Morado. Another restoration assay was established in Valle Morado and Los Naranjos.

In 2001, a 50 ha experimental plantation was established in Valle Morado (Salta Province) that includes 20 native and 11 exotic tree species (**Table 5.1**). A plantation was established with 20 native species planted at random to simulate natural regeneration in order to generate information about the restoration process in degraded or deforested areas. In this assay, no interventions (e.g. pruning) were conducted in order to avoid modifying natural establishment and growth patterns. In the plantation, pure and mixed assays with exotic and native tree species were established to evaluate growth rates, growth form, health, and species interactions (e.g. competition, nurse effect). Treatments included variations in tree density, mixtures of different native species, use of exotics as nurse trees. Tree performance was evaluated in terms of diameter and height increments and vulnerability to pathogens under different treatments.

Surveys of the experimental plantation were performed in 2003, 2005, and 2007. In October 2005, the experimental plantation was damaged by fire (ca. 12 ha); then, during the summer of 2007, 5 ha were replanted with native and exotic species, but unusually low temperatures recorded during the first winter after replanting resulted in the massive loss of seedlings. In March 2008, a 2.6-ha experiment of native and exotic species was established in Valle Morado to take the place of the experimental area damaged by fire and frost. One native species (*Cedrela balansae*) and two exotic species (*Toona ciliata*, *Tectona grandis*) were used in this experiment. In the 50-ha experimental plantation, after eight years of establishment total density was 744 individuals/ha and mortality was 18.5%. Mean diameter for all species was 11.4 cm, mean height was 8 m, and mean basal area was 10 m^2/ha. A multispecific block with high canopy cover was reached after eight years of experimentation with *C. balansae*, *Pterogyne nitens*, and *Tabebuia impetiginosa* as dominant species in the block.

Pure and mixed plantations of native trees displayed different diameter, mortality, and biomass than mixed plantations with exotic species. The native species with the largest diameters were *Enterolobium contortisiliquum*, *Tipuana tipu,* and *C. balansae*, which reached diameters of more than 10 cm in the first five years; whereas the species with the smallest diameters (6–8 cm) were *Cordia trichotoma, Jacaranda mimosifolia, P. nitens, Astronium urundeuva,* and *T. impetiginosa* after eight years (**Table 5.7**). In mixed plantations of native and exotic trees, the diameter and height of native species were smaller to those of exotic species. In contrast, the mortality of native species was lower (between 5–30%) than that of exotic species (>30%). The biomass estimation for exotic tree assays (40 to >70 tonne/ha) tended to be greater than for native plantation assays. Pure plantations of the native tree *C. balansae* (80 tonne/ha) or *T. tipu* (50–60 tonne/ha) reached greater biomass than any other native species during the eight years of evaluation.

Table 5.7 Mean diameter (dbh), height, and annual increase (IMA) for the tree species used in the restoration essay with mixed native species in northern Argentina. All species were planted on January 2002.

Species	*dbh (cm)*	*IMA (cm/yr)*	*height (m)*	*IMA (m/yr)*
Cordia trichotoma	5.78	1.54	3.39	0.91
Cedrela balansae	6.85	1.83	5.54	1.48
Tipuana tipu	12.77	3.41	8.91	2.37
Pterogyne nitens	5.46	1.46	4.76	1.27
Jacaranda mimosifolia	6.88	1.83	4.45	1.19
Tabebuia impetiginosa	6.22	1.65	3.33	0.89
Astronium urundeuva	7.69	2.05	4.66	1.24

Conclusions

(1) Exotic tree species displayed a higher mortality than native tree species, particularly during the dry season (June–October); (2) native species tended to form more ramifications than exotic species; it is therefore advisable to favour high density plantations of native tree species; (3) mixing exotic (*Grevillea robusta*) and native species reduced ramification; (4) *Phylostilon rhamnoides* (native species) displayed low survival and growth in open plantation, owing to its shade requirements for recruitment; (5) some exotic species (*Flindersia xanthoxila*, *F. australis*, *Khaya senegalensis,* and *Paulownia fortune*) proved to be poorly adapted to local conditions; (6) *C. balansae* (native species) suffered severe damage from a shoot boring moth (*Hypsipyla grandella*); chemical pest control and pruning for stem conduction is therefore recommended; (7) *T. tipu*, *C. trichotoma* (native species) and *T. ciliata* (exotic species) displayed the best performance; (8) *T. ciliata* ('cedro australiano') is considered equivalent to *C. balansae* ('cedro Oran'). 'Cedro australiano' is not attacked by native pests and its genetic selection confers certain advantages from a forestry viewpoint, but some assays with *C. balansae* achieved higher biomass accumulation rates than in other secondary forests.

Southwestern Argentina

The study was conducted in the forest-steppe ecotone on the eastern slopes of the Patagonian Andes, Argentina (**Fig. 5.6**). The region is characterized by an abrupt west to east decrease in precipitation owing to the orographic effect of the Andes. Droughts in concert with natural and anthropogenic ignition sources make fire the main driver of ecological change in the region. The high fire frequency during aboriginal and European settlement eras has led to forest fragmentation (Chapter 3); local extinctions of fire-sensitive arboreal taxa and/or the retraction of these taxa into fire-free rocky refuges with low fuel loads have permitted the survival of scattered remnant trees. Treeless areas have been traditionally viewed by foresters and land managers as barren lands unable to support native forest and have been used for extensive sheep, cattle, and goat ranching. In addition, other introduced herbivores such as European hares, rabbits, and exotic deer have a negative impact on native arboreal vegetation. Another source of dry native forest degradation is the current increasing trend of exotic plantation establishment (mostly pine), which drastically changes fire regimes by increasing the extent and severity of fires. Restoration trials were established at two sites located in the central region of the *Austrocedrus chilensis* distribution area, in Arroyo del Medio Valley and in Estancia San Ramón, both privately-owned lands on the outskirts of Nahuel Huapi National Park.

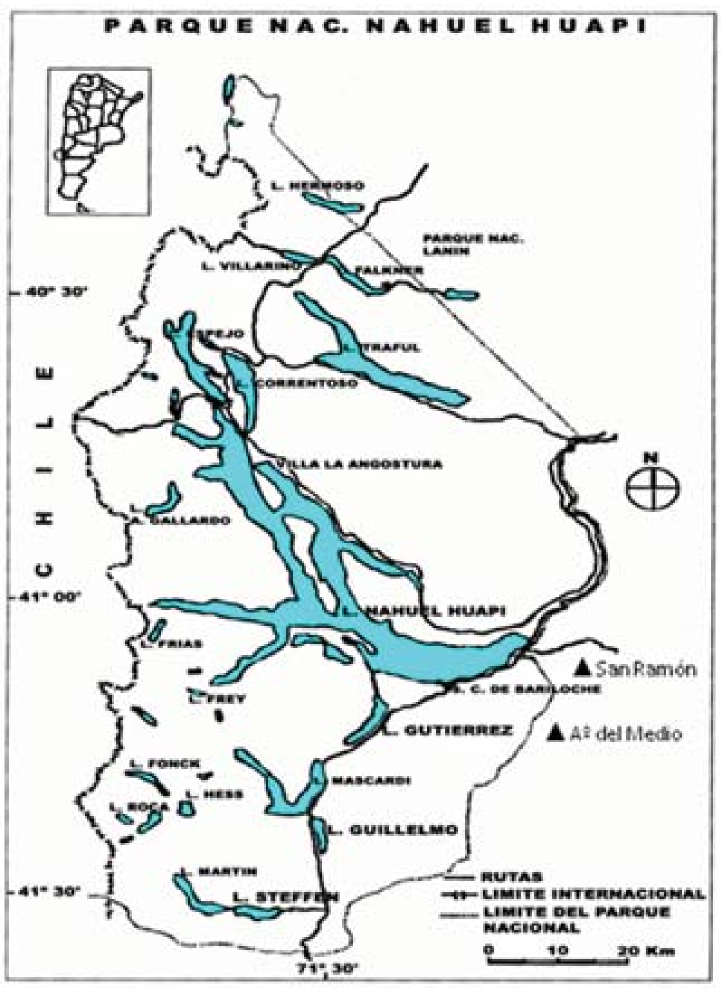

Figure 5.6 Study site in south western Argentina. Experiments were established on two privately-owned lands: Arroyo del Medio Valley and Estancia San Ramón.

Restoration experiments focused on the economically important tree species *Austrocedrus chilensis.* Experiments were designed to test the following hypotheses: (1) *Austrocedrus* is absent from areas where it has become locally extinct owing to higher fire frequency resulting from anthropogenic activities, and (2) *Austrocedrus* establishment is regulated by micro-site availability (nurse plants), herbivory, and traits relating to local adaptation (origin). Experimental trials emphasize the effect of nurse plants (with and without protection), herbivory (with and without exclosure), location on the slope (the highest part of the hill and mid-slope), and proximity to shrubs and grasses (underneath a nurse plant and far from all neighbouring plants).

The first experimental trial was established in Arroyo del Medio during the austral autumn (humid season) in May 2008. *Austrocedrus* seeds were stratified and sown in pots in a greenhouse, and seedlings were transferred to experimental plots. The experiment consisted of: (1) using nurse plants (seedlings planted underneath a shrub or without shrub protection), and (2) herbivore exclosure (with and without). Seedlings with nurse plants and herbivore exclosures suffered very high seedling mortality during the first winter of 2008. Mortality was over 90% in open plots for both seedling cohorts, probably caused by root uplift by frost owing to the small size of seedlings, especially those less than one year old. Nevertheless, survival was significantly higher on treatment plots with nurse shrubs (between 35 and 75% for the very young ones and the 3-year-old seedlings, respectively; **Fig. 5.7**). Herbivores had no effect on survival rates during winter. Owing to slow *Austrocedrus* growth rates, monitoring was only based on survival rates.

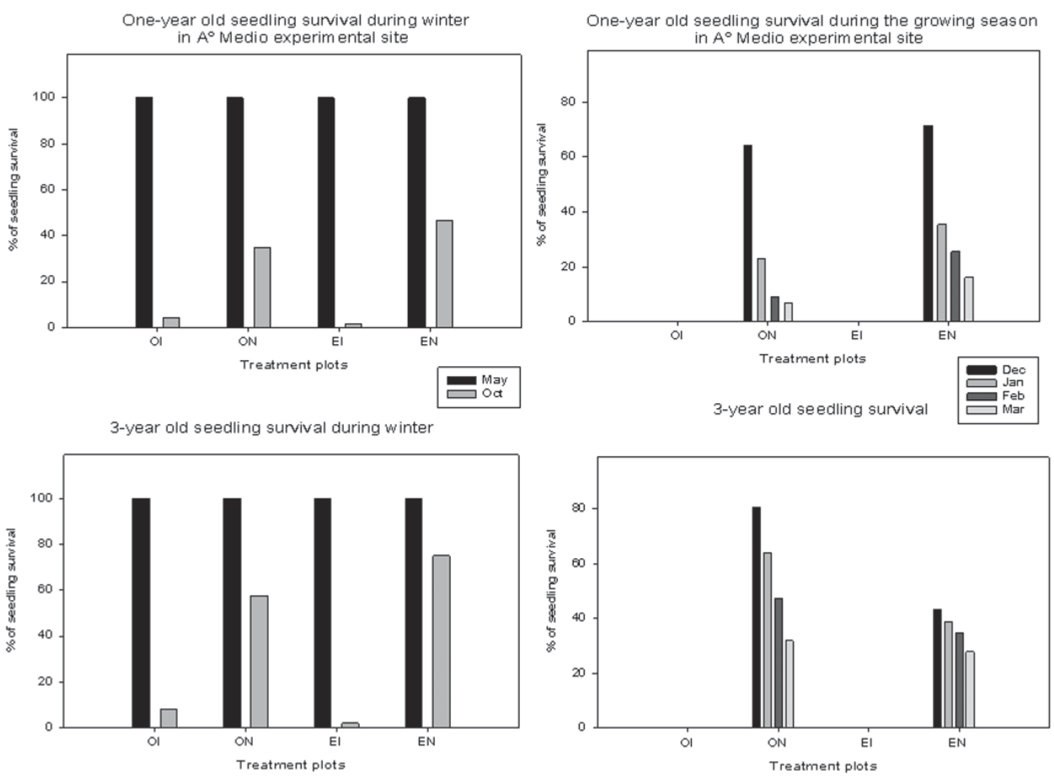

Figure 5.7 *Austrocedrus* seedling survival in Arroyo del Medio experimental site, south western Argentina. Left graphs, seedling survival during the winter under the treatments OI = open and with no nurse shrub, ON = open and with a nurse shrub, EI = exclosure with no nurse shrub, EN = exclosure with nurse shrub. Right graphs, seedling survival during the summer season (December to March) was low in general, even under nurse shrubs (treatments ON and EN), but 3-year-old seedlings had higher survival rates at the end of the season (March). Note that at the beginning of the growing season there were no seedlings alive in plots with no nurse shrubs (OI and EI).

Under the same treatments, survival of 3-year-old seedlings was significantly higher than that of 1-year-old seedlings. After the following growing season, survival was low (between 10% and 30%), probably caused by the lack of precipitation during a particularly dry summer (2008–2009). Only plots with nurse shrubs had surviving seedlings at the beginning of the summer, and there was no significant difference among those with and without protection against herbivores. Again, the 3-year old seedlings had a significantly higher survival rate at the end of the summer: ca. 30% survival during the growing season compared to 10% for very young seedlings (**Fig. 5.7**).

The second experiment was established on San Ramón Ranch during spring 2008 (October), on a small fragment of *Austrocedrus* woodland on a rocky outcrop surrounded by steppe vegetation. Three-year-old seedlings from a commercial nursery were planted in the experimental plots. All seedlings were transplanted underneath a shrub or in artificial shade. Treatments were herbivore exclosure (with and without), location on the slope (on a rocky substrate on the highest part of the hill and in soil on the mid slope), and proximity to shrubs and grasses (underneath a nurse plant and far from all neighbouring plants, covered by a shade mesh). Survival during the growing season was very good (ca. 80%) on the mid slope, in the matrix of grasses and shrubs, in treatments under nurse shrubs, as well as in those with artificial shade. No effects from neighbouring grasses and herbs were detected, at least after one growing season. From November to January, there were no significant differences in survival among treatments. Conversely, survival on the ridge and on a rocky substrate was extremely low in February at the end of summer (**Fig. 5.8**). This may be attributable to a lack of sufficient organic soil to retain humidity; it was also a very dry summer, and the nurse plants used were smaller and less dense shrubs than those on slope plots.

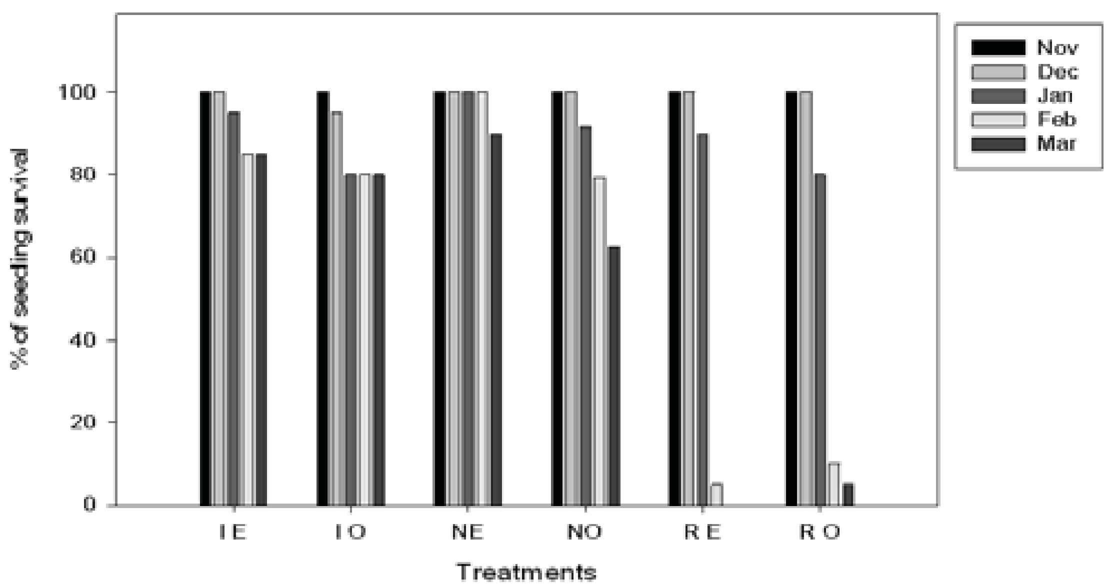

Figure 5.8 Seedling survival among the different treatments was high until mid-summer (January), but towards the end of summer (February and March), survival was almost nil on the rocky ridge (RE and RO).

Conclusions

The main limiting factor for establishment of *Austrocedrus* is drought, especially when it is adjacent to the steppe, where some scattered, isolated woodlands of this species occur. For restoration programmes, it is recommended to use nurse shrubs (to protect against extremely cold weather and soil frost during the winter) or artificial shade (to lessen direct sunshine and wind desiccation during the summer), to improve seedling survival. Seedlings should be planted when they are over two years old, since in their first year they are vulnerable and mortality is very high.

Central Veracruz, Mexico

In central Veracruz, Mexico, the study area was located in Paso de Ovejas and Emiliano Zapata (19°17'N, 96°26'W, 100–250 m a.s.l.; **Fig. 5.9**). The climate is characterized as hot and dry. Mean minimum and maximum temperatures are 20 and 31°C, respectively. Mean annual precipitation is ca. 900 mm and is unevenly distributed throughout the year. The dry season extends from October to May. Soils are mainly cambisols and vertisols with considerable extensions of exposed rock. The land is mainly used for cattle ranching, typically on relatively small-scale private farms. Activities on communal lands are more diversified, the main activity being maize production. Other crops are papaya, bean, green chili, watermelon, sorghum, sugarcane, and mango. Selection of the study area was based on ecological criteria and historical factors. This area has numerous remains of pre-Hispanic settlements (600 to 1500 AD) and played an important role in Mexican independence (19th century). Human occupation of the region has been considerable, as documented by the presence of more than 100 archaeological sites; paintings on cave walls and ceilings indicate pre-Hispanic settlements as well as remnants of the Royal Road, bridges, and estates (*haciendas*) from the 19th century. In this region, five early secondary successional sites with differences in land-use history and time since abandonment were selected to establish the restoration experiments.

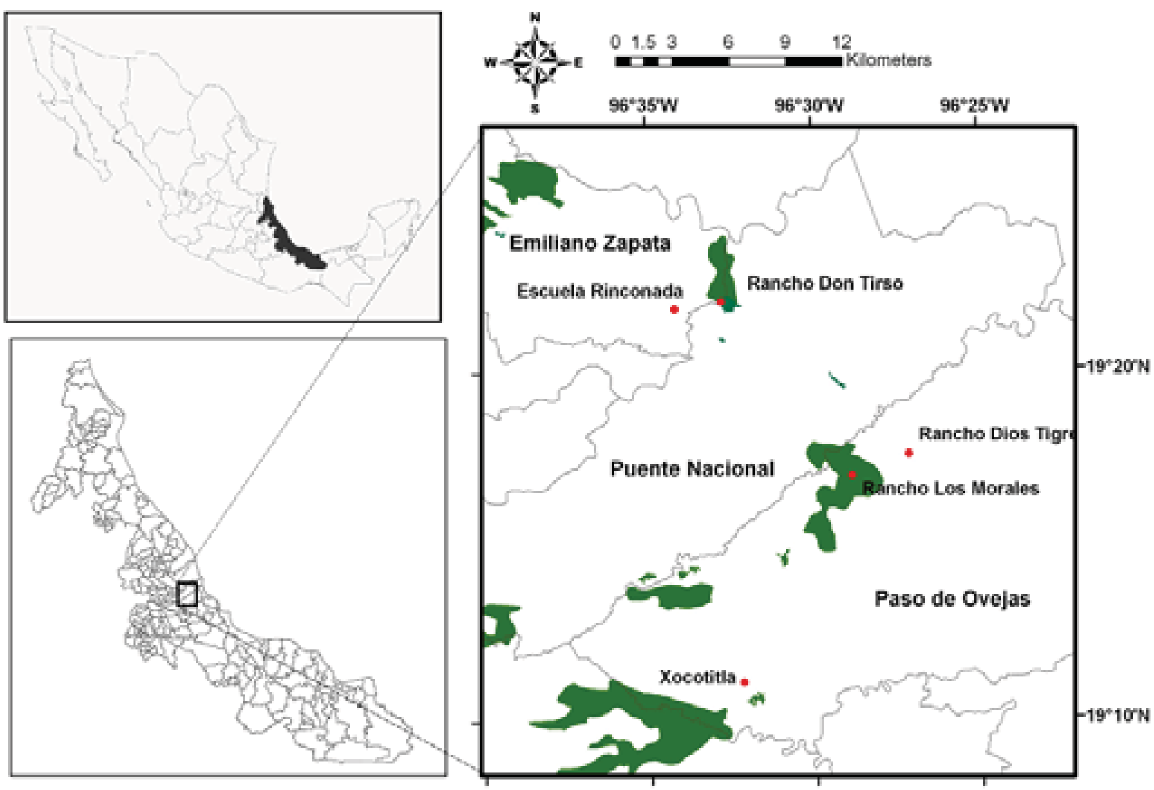

Figure 5.9 Map of the study area in central Veracruz, Mexico. The restoration experiments are located in the municipalities of Paso de Ovejas and Emiliano Zapata, and the sites are Rinconada, Don Tirso, Dios Tigre, Los Morales and Xocotitla.

(1) *Land-use history and site effect on mixed plantations.* Restoration assays were designed to evaluate the potential of selected tree species to establish and grow in fallows with different degrees of disturbance. Four restoration assays were established using six native tree species (**Table 5.1**). In September 2007, a total of 960 seedlings (60 per site, 10 of each selected species) were transplanted to each site. Plant survival and growth in basal diameter and height were then monitored every four months until October 2009 to include two dry and two rainy seasons.

Results indicated that seedling survival was statistically similar among species, with the exception of *Cedrela odorata*. After the first dry season (five months without precipitation), survival of all species was greater than 55%. However, after the second dry-season, survival was the lowest for *C. odorata* and the highest for *Ceiba aescutifolia* and *Guazuma ulmifolia* (**Fig. 5.10**). Following drought periods, *Tabebuia rosea, Ipomoea wolcottiana*, and *Luhea candida* featured an average survival rate lower than 50%. The relative growth rate in height (RGRh) was statistically similar among species, whereas growth in diameter (RGRd) differed (**Table 5.2**). *C. odorata* had the highest RGRd, followed by *G. ulmifolia*, *T. rosea*, and *I. wolcottiana*. *C. odorata*, in spite of its excellent growth performance, appeared unable to adapt to drought conditions. *G. ulmifolia* maintained a high survival percentage owing to re-sprouting of apparently dead individuals during the rainy season. *C. aesculifolia* showed a slow growth because its stem top frequently broke during the dry season, recovering when the rainy season began. For all species, there was a trend toward better performance in old fields with less disturbance and greater age since abandonment. Site differences are the result of previous land uses (cropland and cattle pastures) and time since abandonment (between 1 month and 5 years before experiments started).

Table 5.2 Relative growth rate in height (RGRh) and diameter (RGRd) for tree species used in restoration assays in the tropical dry forest region of central Veracruz, Mexico, during the rainy and dry seasons (2007–2009). Values are mean and one standard error.

Species	RGRh (cm/cm/mo)		RGRd (mm/mm/mo)	
	Rainy	Dry	Rainy	Dry
Cedrela odorata	0.24 (0.03)	–0.02 (0.02)	0.13 (0.027)	0.02 (0.007)
Ceiba aesculifolia	0.05 (0.007)	–0.006 (0.01)	0.05 (0.004)	0.002 (0.003)
Guazuma ulmifolia	0.18 (0.03)	–0.03 (0.01)	0.10 (0.01)	0.01 (0.003)
Ipomoea wolcottiana	0.18 (0.02)	–0.02 (0.005)	0.11 (0.01)	0.008 (0.005)
Luehea candida	0.04 (0.01)	–0.016 (0.007)	0.04 (0.01)	0.007 (0.003)
Tabebuia rosea	0.09 (0.02)	0.01 (0.007)	0.06 (0.01)	0.023 (0.007)

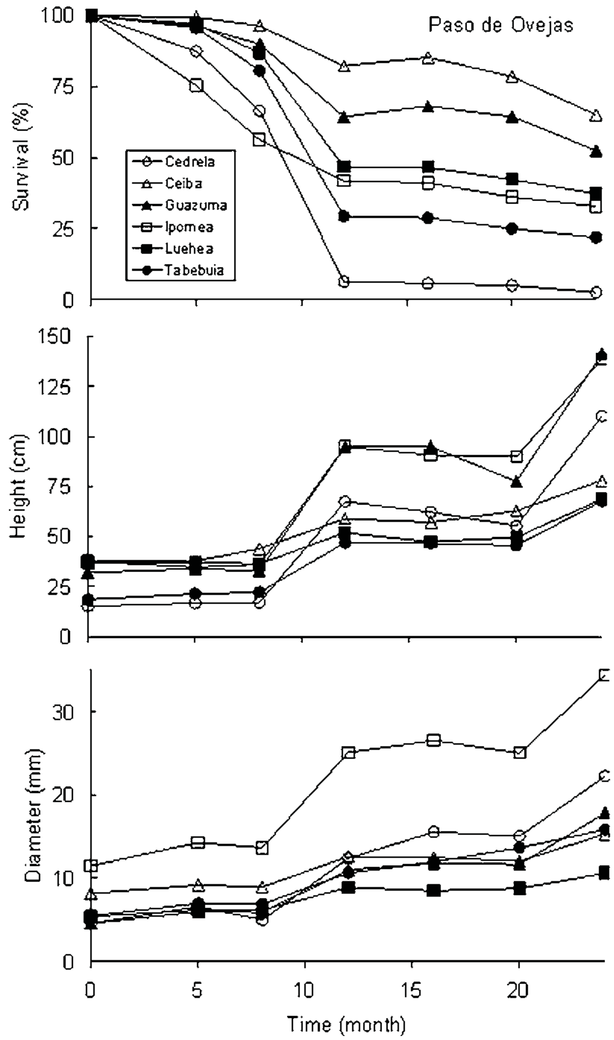

Figure 5.10 Survival percentage, growth in height and diameter of six tree species used in the restoration sites in central Veracruz, Mexico.

(2) *Performance of species selected by local people.* An experimental mixed plantation with eight woody native species selected by local people (**Table 5.1**) was established in the 2008 rainy season for evaluation of initial seedling performance. *C. odorata* was also evaluated in an enrichment plantation. Seedlings of these species were produced in local backyards and transplanted to the field (Xocotitla, 48 months fallow). Survival, growth (height and diameter), re-sprouting, and herbivory (leaf and stem) were evaluated every four months until October 2009. Stem girdling in *C. odorata* and leaf necrosis in *Maclura tinctorea* were also recorded.

The mixed-species plantation displayed a survival rate of 95–100% in five months; however, following the 2009 dry season, survival decreased to 83–97% for most species except *C. odorata*. *C. odorata* had survival rates of 14 and 28% in mixed and enrichment plantations, respectively. In 2008, RGRd was statistically different among species; the highest growth was exhibited by *Lysiloma acapulcense*, *Chloroleucon mangense*, and *C. odorata*. However, for the next dry season (2009) there were no differences among species. In both periods, there were no differences in RGRh among species.

Cedrela odorata performance between mixed and enrichment plantations was similar in terms of RGRd, but RGRh was higher for the former (**Table 5.3**). There, after initial transplanting, girdling stem was the main cause of mortality; during the 2008 rainy season, this type of damage was observed in 11% of individuals that died during 2009. In the enrichment plantation, girdling was present in only 2.3% of the individuals. Re-sprouting was observed in four species (*C. mangense, L. acapulcense*, *Leucaena lanceolata* and *C. odorata*). Damage by herbivores was low in *Cordia alliodora, L. acapulcense, C. odorata*, *Diphysa carthagenensis*, and *Lysiloma divaricatum.* The results suggest that *C. odorata* can be used to enrich secondary vegetation, while the other species have the potential for use in mixed plantations to restore disturbed areas.

Table 5.3 Relative growth rate in diameter (RGRd) and height (RGRh) in eight tree species in mixed and enrichment (E) plantations. Values estimated up to winter (locally called 'Norte') 2008 and dry season of 2009 in Xocotitla, Veracruz, Mexico. Values are mean and one standard error. Means in the same column accompanied by the same superscript do not differ significantly at alpha = 0.05.

	RGRd) (mm mm^{-1} month^{-1})	*RGRh) (cm cm^{-1} month^{-1})*		
Species	*Norte 2008*	*Dry 2009*	*Norte 2008*	*Dry 2009*
Chloroleucon mangense	0.122 (0.02)^{a}	–0.008(0.01)^{a}	0.051 (0.02)^{a}	0.044 (0.01)^{a}
Diphysa carthagenensis	0.085 (0.03)^{ab}	–0.016(0.03)^{a}	0.028 (0.03)^{a}	–0.016 (0.05)^{a}
Lysiloma acapulcense	0.130 (0.05)^{a}	0.001(0.01)^{a}	0.004 (0.03)^{a}	0.037 (0.01)^{a}
Lysiloma divaricatum	0.075 (0.04)^{ab}	0.001(0.05)^{a}	0.050 (0.06)^{a}	0.021 (0.03)^{a}
Leucaena lanceolata	0.059 (0.03)^{ab}	0.010(0.01)^{a}	0.021 0.02)^{a}	0.011 (0.02)^{a}
Cordia alliodora	0.021 (0.03)^{b}	0.015(0.03)^{a}	–0.007 (0.0)^{a}	–0.017 (0.07)^{a}
Maclura tinctorea	0.073 (0.03)^{ab}	0.016(0.01)^{a}	0.042 (0.02)^{a}	0.002 (0.02)^{a}
Cedrela odorata	0.110 (0.04)^{a}	0.003(0.02)^{a}	0.015 (0.00)^{a}	0.046 (0.02)^{a}
*Cedrela odorata (*E)	0.151 (0.02)	–0.023(0.03)	–0.002 (0.00)	0.054 (0.030)

Conclusions

All selected species can potentially be used for restoration; however, an appropriate transplanting site must be chosen for each particular species. Timber species (*C. odorata* and *T. rosea*), which are the species most valued by local people, are severely drought intolerant. Therefore, they have to be planted under nurse trees that help to maintain humidity. In contrast, *C. aescutifolia* and *G. ulmifolia* are drought tolerant and may be used in disturbed areas with no woody vegetation; their re-sprouting ability is an advantage that allows establishment and survival in extremely dry conditions. *I. wolcottiana* and *L. candida* can be used in disturbed areas.

Local people are not enthusiastic about non-timber species, but these are important for restoration because they can change the microenvironment, providing suitable conditions for establishment of other species of economic importance. Enrichment planting is recommended in early successional sites that lack both non re-sprouting and key primary tree species (e.g. *C. odorata* or other multipurpose, foraging, and fruit species). In mixed plantations, *C. odorata* is one of the most valuable tree species in the area; it can be selected if irrigation is available and the density of transplanted seedlings is low.

It is clear that restoration success is directly related to site conditions, since the best species performance was recorded in the least disturbed sites with isolated shrubs or trees that act as nurse individuals. Transplantation of seedlings to disturbed sites is an important technique to accelerate restoration. Selection of tree species for restoration should take into account local knowledge and site characteristics.

Oaxaca, Mexico

The study area is located in the Mixteca Oaxaqueña and Central Valley regions. Experiments were established in the municipalities of Asunción Nochixtlán, Santo Domingo Yanhuitlán, San Pedro and San Pablo Teposcolula in the Mixteca region (**Fig. 5.11**). The area is divided into three precipitation zones (annual precipitation of more than 900 mm, between 600 and 700 mm, and less than 550 mm). The vegetation types are oak forest, pine forest, grassland, and shrubland. In the Mixteca region, plant cover is less than 25%, with dispersed patches of shrubs growing on hilltops (Cruz-Cruz, 2005). Soil erosion is present in 59% of the area with a soil loss rate of over 50 tonne ha/yr (Romero *et al.,* 1986). Indigenous groups (Mixtec) inhabit this region; their livelihood is primarily based on traditional forms of agriculture and the use of natural forest resources. However, this region represents one of the most extreme cases of environmental degradation of drylands in Mexico. The main causes of degradation include high livestock density, agricultural expansion, unsustainable rangeland management practices, and high deforestation rates.

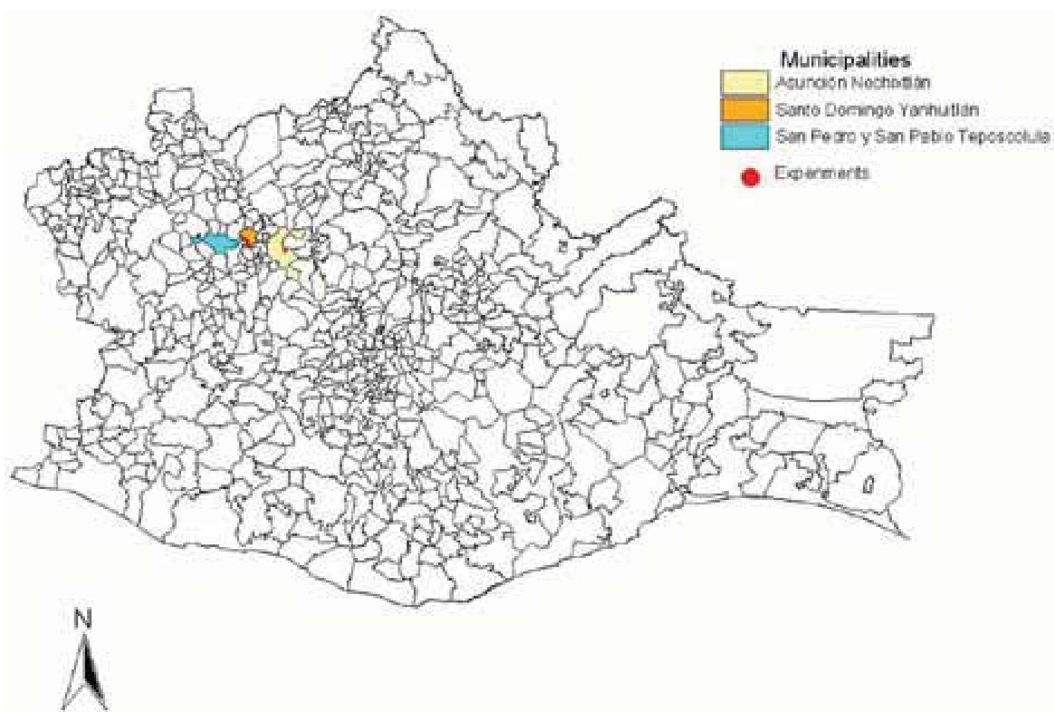

Figure 5.11 Map of the study area in Oaxaca, Mexico. The study sites are located in the municipalities of Asunción Nochixtlán, Santo Domingo Yanhuitlán and San Pedro y San Pablo Teposcolula in the Mixteca Oaxaqueña region.

(1) Native shrubs in four substrate types. The field experiments focused on the performance of native shrubs in four substrate types. The studied species appear in **Table 5.1**. The treatments differed in the soil characteristics of the four study sites. The substrate types were as follows:

- sedimentary red soil (Yanhuitlán Formation, Montmorillonite clay), pH >8.0, organic matter (OM) 1.2%, carbonates 31.9%, and high erosion
- sedimentary limestone soil (white soil), pH >8.0, OM 1.3%, cemented layer by carbonates, and shallow soil
- volcanic tuff soil, pH 6.8, OM 1.9%, low carbonate content, and shallow soil
- black soil (volcanic soil from andesitic material), pH 6.8, OM 6.1%

Field experiments were established in July 2008 (rainy season) and were monitored during August and November 2008 and May and November 2009. The recorded variables were survival, height, cover, and biomass growth rate.

The survival of native shrubs decreased during the 15-month evaluation. *Desmodium grahamii* showed over 95% survival in all substrates and *Acacia angustissima* over 90% survival; whereas over 80% of *Eysenhardtia polystachya* and *Dodonaea viscosa* survived. *Amelanchier denticulata* and *Cercocarpus fothergilloides* displayed lower survival in black soils and volcanic tuff soil. Both collections of *A. denticulata* had only 37% plant survival in black soil. *C. fothergilloides* survived at a rate of 75% in black soil and 70% in volcanic tuff soil. A general trend was that species had lower survival in black soil due to the type and content of clay (20%) and in volcanic tuff possibly due to the high sand content (>55%). During the dry season (November to May), the black soil cracked and the root system was damaged by the expansion and contraction of the soil.

In the four substrates, plant height for *D. orbiculare*, *E. polystachya*, and *D. viscosa* was superior to that of *A. denticulata*, *A. angustissima*, and *C. fothergilloides*. Among substrates, in volcanic tuff and black soil plants grew more than in Yanhuitlán formation and limestone soil. This difference relates to the organic matter and nutrient content. The species included in the experiment are capable of re-growth, as observed at the four study sites. Plant growth rate per week differed among substrates. *C. fothergilloides* reached the highest value in poor nutrient content limestone soil yet showed no nutrient deficiency; consequently, *C. fothergilloides* can be recommended for this soil type. *D. orbiculare*, *A. angustissima* and *E. polystachya* showed high plant growth rates in volcanic tuff and *D. orbiculare*, *A. denticulata*, and *C. fothergilloides* in the Yanhuitlán formation (**Table 5.4**). The results suggested that the species showed different responses in each soil type and probably displayed certain adaptations to these respective environments.

Table 5.4 Mean of growth rate in height and standard error (SE) in four different substrates in field conditions for studied species, Oaxaca, Mexico. n is number of individuals.

Species	Limestone soil Growth rate (cm wk^{-1})	n	Volcanic tuff Growth rate (cm wk^{-1})	n	Black soil Growth rate (cm wk^{-1})	n	Yanhuitlán formation Growth rate (cm wk^{-1})	n
Desmodium orbiculare	0.14(0.04)	12	0.62(0.09)	27	0.30(0.06)	18	0.76(0.50)	6
Eysenhardtia polystachya	0.25(0.08)	15	0.33(0.08)	22	0.16(0.06)	13	0.20(0.05)	23
Dodonaea viscosa	0.34(0.04)	28	0.12(0.01)	35	0.25(0.02)	38	0.26(0.05)	25
Amelanchier denticulata Y	0.24(0.02)	30	0.15(0.02)	26	0.07(0.01)	12	0.24(0.02)	36

Table 5.4 (cont.)

Species	Limestone soil Growth rate (cm wk^{-1})	n	Volcanic tuff Growth rate (cm wk^{-1})	n	Black soil Growth rate (cm wk^{-1})	n	Yanhuitlán formation Growth rate (cm wk^{-1})	n
Amelanchier denticulata T	0.37(0.04)	34	0.15(0.02)	23	0.15(0.03)	12	0.48(0.02)	32
Acacia angustissima	0.23(0.06)	7	0.49(0.13)	8	0.13(0.05)	5	0.29(0.11)	8
Cercocarpus fothergilloides	0.65(0.07)	24	0.27(0.04)	22	0.15(0.03)	21	0.43(0.04)	12

(2) *Nursery conditions*. The same species and the four substrates mentioned above (plus a control) were grown in nursery conditions. On March 2008, seeds were sown in plastic bags. The recorded variables were days to emergence, number of stems, plant height, plant cover, and dry matter per plant. Native shrubs showed significant differences in several variables (**Table 5.5**). The number of days to emergence was different among species. Three species emerged in less than 17 days, while *A. denticulata* emerged after 60 days. *D. orbiculare* developed more stems per plant (2.4) than *C. fothergilloides*, *D. viscosa*, and *A. denticulata* (1.0). The multi-stem production of species at the soil surface level is an advantage for soil erosion control and rain water infiltration. *D. orbiculare* and *A. angustissima* reached the highest plant cover (>270 cm^2/plant), while both collections of *A. denticulata* had the lowest values (<13 cm^2/plant). Plant cover was related to dry matter production per plant. *D. orbiculare* (0.53 g/day) and *A. angustissima* (0.27 g/day) showed the highest growth rates, while the lowest values were recorded for the collections of *A. denticulata* and *C. fothergilloides* (<0.045 g/day) (**Table 5.5**).

Table 5.5 Number of days to emergence, stem number, cover, dry matter and growth rate for studied species in Oaxaca, Mexico. Means in the same column accompanied by the same superscript do not differ significantly at alpha = 0.05.

Species	Emergence (days)	Stems (number)	Cover (cm2/ plant)	Dry matter (gr/plant)	Plant growth (gr/wk)
Desmodium orbiculare	8.90^{a}	2.41^{a}	272.77^{a}	7.23^{a}	0.53^{a}
Acacia angustissima	10.15^{a}	1.11^{bc}	272.63^{a}	4.67^{b}	0.27^{b}
Eysenhardtia polystachya	17.05^{ab}	1.26^{b}	129.38^{b}	2.16^{c}	0.22^{c}
Cercocarpus fothergilloides	18.40^{ab}	1.00^{c}	17.26^{c}	0.35^{d}	0.05^{d}
Dodonaea viscosa	21.20^{b}	1.00^{c}	108.45^{b}	2.87^{c}	0.19^{c}
Amelanchier denticulata T	61.90^{c}	1.00^{c}	11.99^{c}	0.32^{d}	0.04^{d}
Amelanchier denticulata Y	64.55^{c}	1.01^{c}	12.74^{c}	0.41^{d}	0.05^{d}

The number of days to emergence varied for native shrubs planted in different substrates. Seedling appearance (24 days) was faster in black soil and in the control. In limestone and Yanhuitlán formation soils, it took 33 days (**Table 5.6**). Black soil and the control were associated with the production of more than 1.3 stems per plant. For plant cover, dry biomass, and plant growth rate, limestone soil and Yanhuitlán formation soil types limited plant growth. The lowest values of all recorded variables corresponded to Yanhuitlán formation soil (**Table 5.6**). Dry matter production per plant was closely related to plant cover and growth. Dry matter values were higher in black soil and in the control than in limestone and the Yanhuitlán formation, a pattern similar to plant growth rates.

Table 5.6 Type of substrate and variables recorded in Oaxaca, Mexico. Means in the same column accompanied by the same superscript do not differ significantly at alpha = 0.05.

Substrate	Emergency (days)	Stems (number)	Cover (cm^2/plant)	Dry matter (g/plant)	Growth (cm/wk)
Black soil	24.60^{b}	1.33^{a}	228.25^{a}	5.65^{a}	0.36^{a}
Control	24.75^{b}	1.34^{a}	219.54^{a}	5.37*	0.35^{a}
Volcanic tuff soil	27.89^{ab}	1.24^{ab}	67.15^{b}	0.88^{b}	0.13^{b}
Limestone soil	33.32*	1.22^{ab}	42.18^{bc}	0.55^{b}	0.06^{c}
Yanhuitlán Formation	33.82*	1.13^{b}	32.31^{c}	0.41^{b}	0.06^{c}

(3) *Vegetative reproduction*. A vegetative reproduction experiment was established in a greenhouse with two shrub species, *Cercocarpus fotergilloides* and *Amelanchier denticulata.* Each stem was treated with Indol–3-butiric acid 10,000 ppm (Radix 10,000). Sprouting and rooting were harvested in March and evaluated in November 2009. The vegetative reproduction experiments showed that *C. fothergilloides* cuttings did not develop roots in any of the treatments. In contrast, *A. denticulata* cuttings developed roots with the following treatments: 17–33 % in Yanhuitlán collection, from 2.0 cm cutting diameter, and 17% in Yanhuitlán collection, 1.0 cm cutting diameter.

Conclusions

Since the long dry season affects seedling survival, particularly in black and volcanic tuff soils, it is advisable to transplant at the beginning of the rainy season in order to allow the development of a better root system. More research is recommended on seed germination, growth of native tree species, and vegetative reproduction from cuttings.

Conclusions

In view of the wide variety of regional differences in the drylands of America, and the relatively small proportion of area covered in this project, no attempt has been made to produce general recommendations. Nevertheless, the different dryland restoration experiences evaluated in this chapter allowed us to discern the factors influencing successful forest restoration and to produce some specific recommendations. The results of field experiments established in each study area, along with other studies, led to the identification of some key ecological processes that limit the establishment and growth of threatened and/or socioeconomically important native tree species found in dry forests. Furthermore, these experiences permitted the identification of restoration techniques that overcome some of those constraints. Thus, based on the factors that influenced successful restoration, the major findings have been grouped and presented below.

In all of the study areas from Mexico to Patagonia, the main limiting factor for establishment was reported to be drought. This finding may be obvious but is not trivial. Air temperature is not a limiting factor except in south western Argentina during the winter, but the irregularity of precipitation is a problem. Restoration efforts in the drylands of America have to confront a common environmental constraint, namely the long dry season affecting seedling survival during any transplanting effort. In order to permit the development of a better root system, transplantation should take place at the beginning of the rainy season – should be planned in accordance with the incidence of El Niño rainy years. Results in different study areas suggest the use of supplemental irrigation for plantations established on open sites during the summer, even when precipitation has been normal that year.

The use of nurse species proved to be important for protecting seedlings from desiccation, thus improving seedling survival and initial growth. In Patagonia, nurse plants provide protection against extremely cold weather and soil frost during winter.

Another technique suggested was the use of terraces to facilitate plant survival. This increases rainfall collection and concentrates moisture at the bottom of the furrow. However, in Oaxaca few communities use them because of their high cost (**Box 5.4**).

Since the effect of herbivores is sufficient to kill most seedlings, the results of experiments in all study areas and under any environmental condition suggest that it is essential to exclude cattle and small mammals before starting a restoration effort. Exclosures were not implemented in only one case – when extended plantations were established.

The use of native tree species was preferred in all study cases. However, employing exotic species is recognized as important in some circumstances, such as in very degraded sites or mixed plantations. It is noteworthy that native tree species displayed higher survival rates than exotic species, especially during the dry season. Preferably, the choice of native species should be based on their drought tolerance as well as on the site disturbance level. In some forests, native species need to be selected because of their re-sprouting ability. On occasions, particular species must be chosen for a given transplanting site; locally important site factors include soil type (e.g. Oaxaca, Mexico) or location on the slope (Patagonia, south western Argentina).

Appropriate methods for restoring dryland forest ecosystems are those that contribute both to the conservation and restoration of biodiversity and to local economic development. It is suggested that seedling transplant restoration techniques should be applied in deforested/degraded areas and seedling transplant enrichment employed in logged forest and old successional sites. Results from most of these field experiments suggested that natural regeneration can be encouraged by protecting successional areas from cattle (exclusions for livestock/small mammals), fire, and selective cutting; enrichment planting is an appropriate method in early successional sites lacking non re-sprouting and key primary tree species; and mixed species plantations can be established on highly degraded sites.

When planning a restoration strategy, it is important to include species of economic and social value as well as endangered species. Local knowledge must be taken into account for the selection of tree species; local people must participate in the selection and be aware of the importance of forest recovery owing to the environmental services that it provides. Information on dry forest native tree species has to be made available to local people through workshops and guided walks. Also, demonstrative restoration assays should be displayed in school yards or communal land for educational purposes. Training in tree species propagation is essential in order to increase the establishment and management of native species plantations.

A lack of knowledge of the biology of native tree species and secondary successional processes limits their implementation in management and conservation plans. Information about native trees is restricted to a few species; it is therefore paramount to carry out plant species research that addresses their phenology, seed dispersal, germination, growth, and vegetative reproduction. Along with biological studies on native tree species, another suggested approach to improving restoration techniques is to learn from studies of secondary succession.

Dryland restoration requires more time than the duration of the experiments described here. However, the experience that has been obtained, based on comparison between and within countries, studying specific cases and different potential solutions, represents a first step in identifying general approaches to the restoration of dry forest ecosystems. From these experiences, we recognize that dryland restoration requires more research since it is not possible to directly apply techniques learned from experiences in other forest types in more humid environments.

***Tabebuia rosea* in central Veracruz, Mexico. Photo: C. Alvarez**

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6 SOCIOECONOMIC VALUATION OF DRYLAND FOREST RESOURCES IN DRY AREAS OF ARGENTINA, CHILE AND MEXICO

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Introduction

Extensive areas in Latin America, arid or semi-arid, are the homeland of millions of people, and have been inhabited since pre-Columbian times (UNDP, 2004). Despite their harsh environmental conditions, these areas are extremely diverse and abundant in plants with actual or potential economic value (Newton, 2008). The value of these plants has been documented for centuries in several classical books such as those of Francisco Hernández (1659) or Maximino Martínez (1936) in México, just to mention a few examples. Despite the long tradition of collecting knowledge about native plants, habitat destruction and soil erosion prevail in many of these areas, constituting a severe threat not only to biodiversity but to the well-being of the local people. Documenting the socioeconomic relationships between local people and native plants is clearly essential for proposing any cogent plan for forest restoration and conservation in these areas. The present chapter summarizes the findings of socioeconomic studies on native dry forest in Latin America and provides recommendations to be taken into account for implementing restoration or conservation plans. The following text summarizes the results obtained from a variety of different socioeconomic surveys conducted in the different study areas as part of the ReForLan project. Further details of the individual studies, together with some information from other relevant research initiatives, are provided in **Boxes 6.1–6.6**.

Results of socioeconomic surveys

Patterns of plant use and indigenous knowledge

Native plant species of dry forests in Latin America can be an important source of products, which can contribute to improve the well-being of local human communities. These products are used in general for subsistence and, to a lesser extent, for commercial purposes. However, many native species that are recognized as having potential uses are not currently being used. In fact, we found that, in some areas, local people are not always aware of all the benefits of native species found in dry forest areas of Latin America.

Awareness of the importance of native plant species of dryland forests varies considerably among regions and even among people within the same region. In central Chile, for instance, very few of the sclerophyllous forest species traditionally known as sources of medicine, food, and fibre were cited in the interviews conducted with local people. These results show that the knowledge of traditional uses of sclerophyllous species by rural inhabitants of central Chile is very limited and has been gradually lost over time. By contrast, in Paso de Ovejas in central Veracruz, Mexico, a region with mainly secondary vegetation and some remnants of tropical dry forest, data compiled from different sources (workshops, in-depth interviews with key informants, field walks with informants, and botanical collections) documented 76 species in one or more categories of use, from primary, secondary, agroforestry, and riparian habitats. In northwestern Argentina, a quantification of the use of medicinal plants in Kolla communities revealed that 117 plant species belonging to 52 families and 98 genera are being used (Hilgert, 2001).

In the dry areas of the Upper Mixtec Region in Oaxaca southern Mexico, a survey of 322 people on the uses of a sample of 112 native local plant species revealed that all species were recognized as useful by at least some of the interviewees. However, only 13 species were judged to be useful by 60% of them; and, on average, only 33% of the species shown were identified as useful to the interviewees. In the sclerophyllous forests of central Chile, a total of 12 species were recognized by local people as being potentially valuable. Overall, the Quillay, *Quillaja saponaria*, is preferred by most of the people with 51.9% preference, and used as source of honey and as a shade plant.

Knowledge about native plants among local people is unevenly distributed among the population and is usually concentrated among very few people. This knowledge is also irregularly distributed even among different municipalities within the same district. In the Upper Mixtec Region in southern Mexico, the studied municipality with the lowest level of formal education, economic welfare, and health services displayed the highest use value of native plant species among respondents. As expected, prevalence of local traditions and culture was strongly related to knowledge of native plants. As formal education is becoming more common and the native language (Mixtec) is being lost, awareness of the importance of native plants species is also being lost. Thus, rescuing traditional knowledge can be an indirect way of preserving native plant species. At a global level, it has been estimated that 80% of the cultural knowledge of indigenous people will disappear in the next one hundred years (Inter-Commission Task Force on Indigenous Peoples, 1997). This cultural loss will certainly jeopardize native forest species since few people will recognize the value of such species. This loss is likely a major driver of species extinction.

It is important to recall that the richness of traditional heritage stems not only from knowledge of a particular use for a particular native plant species, but from the diversity of uses given to such a species. Thus, in surveys assessing traditional knowledge, it is essential to evaluate not only the number of species with a use value for the local people but also the diversity of uses the community gives to native species. For example, in the dry forest of the Upper Mixtec Region, Oaxaca, Mexico, the following uses attributed by local people to native plants have been identified, in declining order of importance: forage for domestic animals, medicine for both domestic animals and humans, a source of energy (firewood and charcoal), and food for humans. Suárez *et al.* (submitted) reported that in Paso de Ovejas, Veracruz, Mexico, farmers identified a total of 12 uses for 76 native species (mostly in the Fabaceae family). Workshop participants agreed that the most important uses in terms of quantity were fuel and fence posts.

Firewood is the main energy source for heating and cooking, and is one of the most common uses of native dry forest (see **Boxes 6.1** and **6.2**). On communal lands, firewood is usually free for local people, but the efforts to replenish firewood extraction in native forests or plantations are nil or insufficient. In the Colliguay Valley, Chile, firewood, charcoal and organic soil are the most commercialized products of dry forests. Yet they have the lowest added value and their extraction has the highest impact on forests. Indeed, they contribute only 6.7% to total family income (Schiappacasse *et al*., 2009).

Recently, in the Upper Mixtec region in municipalities such as Santiago Tilantongo, plantations of oaks have been established in degraded areas. These plantations can help to reduce the effects of harvesting for firewood, but more studies are needed to assess the amount of firewood needed per inhabitant and the capabilities of firewood production of native forests or plantations under different harvesting scenarios. Such studies are now being conducted for some localities such as Ocuilapa de Juárez, Chiapas, where firewood is also used for heating ovens for pottery. Therefore, firewood demand can be very high and the supply of local forest can be insufficient. Clearly, establishment of plantations for energy and analysis of the production of firewood are urgently needed (Holz and Ramírez Marcial, in preparation).

In northwestern Argentina, seasonal dry forests (premontane Yungas and Chaco) provide firewood for most local communities living in or close to these forests. Collection and use are primarily performed on a per-family basis, accessing relatively small amounts each time but consistently throughout the year. For instance, in Los Naranjos, an indigenous Kolla community settled in the premontane forests, a preliminary estimation showed that the community (about 260 inhabitants) uses on an annual basis about 315 trees of different sizes for firewood, and about 500 for fence posts. Hardwood species from the Myrtaceae and Fabaceae families were preferred for firewood, while a wider range of species were used for fence posts, including Quina (*Myroxylon peruiferum*), Afata (*Cordia trichotoma*), Lapacho (*Tabebuia* spp.), Nogal (*Juglans australis*), Mato (*Myrcianthes* spp.), and Cebil (*Anadenanthera colubrina*). A high intensity of firewood use was detected in the forest in a 2 km radius around the community, after which distance the intensity of use decreased sharply. We do not know whether the spatial distribution of firewood extraction in the forest has been stable over time. The community settled permanently in the mid 1980s, and no information relating intensity of use to forest growth is available.

Tree species with high potential for restoration

In most parts of central Chile, native dry forests are subjected to intense logging to satisfy the internal demands for forest products or to clear cutting for the expansion of agriculture (Newton *et al.,* 2009; Schiappacasse *et al.,* 2009). Boldo *(Peumus boldus*) and quillay *(Quillaja saponaria*) have been identified as an attractive alternative for restoration programmes (Echeverría *et al.,* 2010).

In the Upper Mixtec region, *Juniperus flaccida* was the most common native plant used, as 90% of the informants recognized this plant as useful. This conifer is a promising plant for restoration purposes, since it can grow in highly degraded areas, its wood is of high quality in particular for construction purposes, it is resistant to rotting, and it is very hard and durable. Furthermore, *Juniperus* is considered to be highly resistant to drought and can actively colonize degraded areas (Willson *et al.,* 2008). Unfortunately, growth rate is slow. On the other hand oaks (*Quercus* spp.) are valuable alternative options for restoration as they are very useful as firewood. In particular, the yellow oak (*Quercus liebmannii*) is the most valued firewood species owing to its rapid ignition and its durability as a fuel.

Overview of agricultural and forest landscapes in the Central Depression of Chiapas. Photo: N. Ramirez-Marcial

One of the productive activities of the inhabitants of Ocuilapa, Mexico, is clay pottery, which requires significant amounts of firewood for fuel. Photo: N. Ramirez-Marcial

In central Veracruz, species such as *Chloroleucon mangense, Lysiloma acapulcense, Leucaena lanceolata, Cedrela odorata*, *Caesalpinia cacalaco, Tabebuia chrysantha* and *T. rosea* are used as living fences and for enrichment of native woody vegetation with linear plantations. *Diphysa carthagenensis* and *Gliricidia sepium* are also suitable plants for restoration in this area owing to their multiple uses.

In northwestern Argentina, native species such as *Cedrela* spp., *Tabebuia* spp. and *Tipuana tipu* have been recommended for restoration efforts, owing to their high growing rates and medium to high timber quality, with a direct economic return as timber over the mid-term (20–30 years). Such combinations of features make these species attractive for private and public investment in restoration.

In all of the studied regions, very few of the most important native species are available from nurseries. Therefore, establishment of nurseries for restoration activities is urgently needed.

Potential for forest restoration activities

Socioeconomic analyses conducted in northern Argentina in the Yungas region compared forest activities with other land-use alternatives, such as sugarcane and soy plantations. From an economic perspective, forest activities were much less profitable than the alternatives. Furthermore, of the potential forest species, native species were less profitable than exotics, and required a government subsidy. These analyses, however, did not consider the indirect environmental and social costs of crops, such as biodiversity reduction, water contamination and subsistence needs. Reforestation and forest-enrichment projects in Yungas could be an option to give more value to relatively unproductive lands. Such projects could also offer a way to diversify commercial activities.

Communal and private lands display different opportunities for restoration. In Chile, for example, people are eager to reforest but not on their own land. In Yanhuitlán in the Upper Mixtec region, some severely degraded areas are privately owned, but the owners live abroad, and local authorities and government agencies although eager to reforest these areas cannot conduct any action without the consent of the absent landowners.

One of the most important benefits of forests to local communities is the ecosystem services that they provide. Recognition of these services could provide a powerful incentive for forest restoration actions. In Paso de Ovejas, Veracruz, the local population is highly aware of the varying functions of trees in the landscape, and values these species accordingly. In the Upper Mixtec region, owing to a chronic scarcity of water for the communities, local populations with the financial aid of federal government programmes are reforesting several areas with the aim of improving water infiltration and reducing runoff from hillsides. Reforestation can be identified even using remote sensing tools. Nevertheless, vast areas of the Upper Mixtec region remain highly eroded, and water scarcity still prevails in the region and is blamed in part on the high rates of forest destruction. Furthermore, reforestation actions have been commonly conducted using exotic species such as *Casuarina* sp. and *Pinus* spp., the long-term effects of which are unknown; local people do not recognize these species as useful. Frequently, reforestation activities are badly planned and lead to the expensive introduction of exotic species that far from helping, have questionable benefits for ecosystem function and conservation, and for the well-being of local communities.

ECOSUR conducted a socioeconomic valuation and market analysis of dryland forest resources in the southern state of Chiapas, Mexico, in the central region. This region does not have a formal market for forest products, although illegal extraction is common. Local stores and carpenters add value to forest products, but all the markets are small in scale. Demand is higher than supply and people in this region import forest products from other regions. Firewood is an important forest product, but it is vulnerable to overexploitation because of high demand from the craft-making sector. The long lead time required to obtain commercial benefits from forest products discourages forestry as a profitable activity for many landholders. Instead, cattle and sheep ranching are commonly preferred. Different government programmes lead to contradictory impacts on native forest, supporting forestry activities on one hand and ranching and deforestation on the other. Such programmes have an important influence on the decisions made by landholders regarding their economic activities, resulting in a complex socioeconomic and political context for forest restoration in the central region of Chiapas.

Forest activities compete with cattle ranching and traditional cropping in virtually all surveyed areas. The agriculturally most valuable areas are usually used for economic activities other than forestry, which, in turn, is relegated to low quality sites such as steep sites or areas with poor soils. Good forestry practices in these low quality sites, however, can help improve the well-being of the local people by providing basic needs such as firewood, and ecosystems services such as enhanced water infiltration and soil retention to prevent soil erosion.

Conclusions

Dryland forest species in Argentina, Chile, and Mexico can be an important source of economic resources to local people and provide essential ecosystem services at the same time. Nevertheless, rates of forest loss and degradation are often high and jeopardize the existence of native forest species. The use value of many such species has been identified through interviews and workshops with local people. Although many native tree species of dryland forest in the study areas were recognized as useful, knowledge about the use value is being lost and, at least in some cases, it is unevenly distributed among local people. Furthermore, formal commercialization of such products is uncommon. Firewood is one of the most widespread uses in all the study areas, but, in general, actions aimed at replenishing the losses due to extraction are either non-existent or insufficient. We conclude that socioeconomic factors are major drivers of habitat destruction and deforestation in the studied drylands. These include the loss of awareness of the importance of native forest species among local people, disengagement of formal education from local knowledge and traditions, insufficient information on the potential economic or ecological importance of native plants, lack of commercialization channels for native forest products, conflicting governmental policies, the introduction of exotic species, and a lack of coordination among stakeholders involved in forest management and conservation. Based on a consideration of these problems, a series of recommendations are provided.

1. Awareness of the importance of dry forest as a source of goods and services for humans is being lost as traditional knowledge is supplanted by formal education. Thus, rescuing traditional knowledge is important for demonstrating the use value of native plants and to provide a motivation for preserving and restoring populations of native plant species
2. Formal education is disengaged from traditional knowledge and tends to ignore tradi-

tional values, including the native language and the use value of native plants. Therefore, including local culture and traditions in formal education would indirectly support the conservation and restoration of native forest.

3. Native plants and forests are commonly overexploited, particularly for firewood (Oaxaca, Chiapas, Chile). Increased forest restoration actions are urgently needed in the study areas to counterbalance the current high rates of firewood exploitation.

4. Nurseries with local native plants are rare or absent in most cases. This is one of the reasons why reforestation programmes use exotic species in some regions of Latin America. Therefore, nurseries with local plants should be encouraged in all areas.

5. The lack of market channels for trading native forest goods and services is one of the reasons why native forest resources usually have little or no monetary value. In many cases local people do not receive a clear economic benefit for preserving or restoring native forests. Indeed, in Oaxaca and central Chile, formal commercialization of forest resources is uncommon. Commercial use of native forest plants, although less profitable than that of exotic species in some areas, can be an option for economic diversification, with lower economic and ecological costs than other economic activities. Exploitation of native dryland tree species can in some situations be profitable, such as the case of boldo in northern Chile. Similarly, timber extraction in northern Argentina is economically important for many small to medium private companies and rural families. The current challenge is to identify, disseminate and adopt sustainable forest management practices and to develop local manufacturing to generate *in situ* welfare and working opportunities.

6. Government policies clearly play an important role in preserving, restoring or destroying the forest. Conflicting governmental policies from different agencies reveal lack of coordination and hinder forest conservation and restoration actions.

7. Government reforestation and restoration programmes should attend to the needs of local people and preferably use native plants instead of exotic species, which may provide little or no economic benefit to the local communities.

8. Native species can have high international market demand as is the case of boldo and quillay in Chile. Clearly, more studies on native plants of dry forests in Mexico and in northern Argentina are needed to explore their potential economic benefits.

9. The importance of ecosystem services should be emphasized among local people and policy makers as an important incentive for conserving and restoring native forest areas in all regions. It is therefore crucial to promote the monetary valuation of ecosystem goods and services provided by dryland forests and to support rural livelihoods (see also Chapter 8).

10. Coordinated participation of stakeholders, namely federal and local authorities, academics, educators, and local people is essential for successful forest restoration programmes.

***Cedrela odorata*, Chiapas, Mexico. Photo: R. Vaca.**

Box 6.1 Firewood consumption for pottery and projections for woody biomass production from *Bursera simaruba*

R. Hernandez, S.C. Holz and N. Ramirez-Marcial

Firewood is the main fuel source for many rural populations in developing countries. In Mexico, fuelwood accounts for 80% of the energy used in rural households. Chiapas is among the five states of Mexico in a critical situation at the junction between consumption and availability of woody energy. Given the importance of fuelwood in the rural Mexican context, there is a need to generate information to support sustainable forest management based on the many tree species used for firewood. Our research aimed to estimate: (i) the woody biomass production of *B. simaruba,* and (ii) the consumption of firewood used in local pottery.

Pottery is an activity practised throughout Mexico and represents an important avenue of artistic expression and cultural transmission in addition to being a means of supporting economic growth of the people involved. In the Central Depression of Chiapas one of the softwood species commonly used for baking pottery is *Bursera simaruba* ('palo mulato'), a native tree species that is widely used in hedges. The work was undertaken in the community of Ocuilapa de Juarez, located in the Central Depression of Chiapas (16°53'52"–16º50'47"N, 93°27'28"–93º 24'17" W). To determine biomass production, we selected 40 individuals of *B. simaruba* of different sizes, and measured the total height, diameter at base (DAB) and diameter at breast height (dbh) of each individual, then determined their biomass through direct harvesting. We estimated firewood consumption during the burning (baked clay pieces) conducted in 13 pottery events.

The dry biomass production increased rapidly during the early years of growth, and the highest values were recorded from 20 years (up to 700 kg per tree). To predict woody biomass from the stem diameter of individuals, we identified a linear equation (r^2 = 0.93, p <0.001). Firewood consumption varied widely among the different pottery burning events (39 to 295 kg), while the median consumption was 81.92 kg. Although many species were used during burning, the most commonly used was *B. simaruba*. From the biomass production data we calculated the number of individuals of *B. simaruba* needed to produce one tonne of dried wood. In a scenario where trees of *B. simaruba* have high biomass production, it would take about 35 trees of 10–20 cm of DAB (aged 4–10 years), while in a scenario with low production of biomass, about 70 trees of the same size categories would be needed to produce one tonne of dry wood. The information generated in this work has been disseminated and discussed with local producers through community workshops. To apply these results, there is a need to support the development of forest plantations of native species for the purposes of providing wood energy, which would require the involvement of local self-government and forest governmental authorities.

Box 6.2. Patterns of firewood use in a tropical dry forest landscape in central Veracruz

M. E. Ramos Vásquez and F. López-Barrera

Firewood plays a significant role in the energy requirements of many developing countries. However, patterns of fuelwood use are determined by a range of social and environmental factors. Moreover, the impacts of firewood use are not necessarily negative for all groups of factors and can lead to permanent environmental change. We explored the variation in patterns of firewood use within the municipality of Paso de Ovejas (see also Chapter 10). Firewood collection and acquisition were compared across two study areas that present different environmental and social conditions. In the upland area (320 m a.s.l.), the predominant land uses are rain-fed agriculture, grasslands and secondary forest, but also in this area the steep topography has allowed the preservation of the largest forest fragments within the municipality. The communities are mainly rural. On the other

Box 6.2 (cont.)

hand, the lowland area (20 m a.s.l.) is dominated by permanent and irrigated agriculture, grasslands and urban areas. The communities are mainly urban and a higher proportion of the inhabitants migrate to work in nearby cities such as Veracruz. Past surveys made by INEGI (Instituto Nacional de Estadistica, Geografía e Informática, 2000) showed that wood fuel dependence of the municipality is relatively low (38% of households). However, these surveys are biased as the presence of a stove in a house does not exclude the use of wood fuels by family members. Therefore, the present study evaluated in detail the patterns of firewood use through social surveys. By extrapolating our data, we estimated the sustainability of current extraction rates.

A total of 136 interviews were conducted in four study sites that were located in both the upland (Acazonica (33 interviews), Angostillo (23), Rancho Nuevo (8), and El Limon (7)) and lowland sections of the municipality (Carretas (12), El Mango (13), Loma Fina (15), and Cerro Guzman (27)). In addition, to explore the environmental perception of the rural communities, a community-wide workshop was divided into two sessions and held on the communal property of the '*Ejido El Limón*'. The first session explored how local habitants perceive dry forests in connection with environmental problems such as degradation, forest transformation to other land uses, and habitat fragmentation. In addition, information was obtained on the value of this type of forest to local inhabitants. The second session was designed to distinguish between the ecological structures of forests denominated '*acahual*' (old-field), '*monte*', and '*mata*', as well as to determine the common tree species present in each type of forest and their potential utility for communities in the study area (including construction, live fences, shelving, firewood, medicine, fodder, and for use in the production of handcrafts). The two meetings lasted approximately two hours each and were initiated by projecting images of the local flora and fauna of dryland forests in an effort to set the stage for later discussions on conservation and sustainable use of this type of forest. **Table 1** describes the pattern of firewood use in the eight localities surveyed. Three firewood units ('*cargas'*) were weighted and their volume was obtained, resulting in an average of 83.7 kg and 0.117 m^3, respectively. We calculated the volume (m^3) of firewood per year consumed in the localities. We found that firewood consumption is significantly higher in the upland communities compared to the lowland communities (p <0.05, **Table 1**).

Using forest structure data (1 ha = 365 m^3 of wood; Williams-Linera and Lorea, 2009), the estimated annual rates of firewood consumption in the municipality (20,787 m^3/year) and the area of secondary and preserved forest recorded in 2007 (1857 and 8672 ha, respectively, see also Chapter 1), we estimated the period of time taken for the forest to disappear owing to this activity. The forest cover loss rate due to firewood extraction resulted in 69.3 ha/year, therefore the period of time that the preserved and secondary forest will last considering the calculated firewood extraction rate within the municipality is 28 and 125 years, respectively. It is important to note that this estimation does not consider forest regeneration (currently being examined; Manson *et al*., in prep.). It was also found that habitants of the upland area of Paso de Ovejas are willing to plant useful trees, however they do not have enough land available in their properties as these have an average size of 1 or 2 ha. These small pieces of land within *ejidos* are dedicated to agriculture for subsistence needs. We also documented that firewood extraction is related to forest conversion, as it forms part of a land management strategy for landowners who supply trees derived from land-use conversion. In the upland area conversion is mainly from secondary forest to agriculture and grasslands, whereas in the lowland area it is mainly from fruit tree plantations to sugarcane.

This study revealed that at current output levels of firewood consumption, sustainable production is possible, and outlined that the firewood extraction is not the main cause of recent forest loss in the municipality, which is mainly attributable to conversion to other land uses. In spite of this trend, past extraction rates of firewood are thought to be higher than current rates, therefore we have to be cautious in attributing the present forest area and condition in the municipality to the impact of firewood extraction.

Box 6.2 (cont.)

Table 1 Summary of the survey on fuelwood use in four upland and four lowland communities in the study region in central Veracruz, Mexico. *Information from INEGI census (2000).

Localities	* No. of houses	*Firewood use (%)	Firewood units/ year	m^3/year consumed	Tonnes/year
Uplands					
Acazónica	294	84.0	12,034	1,408	1,007
Angostillo	184	77.7	5,285	619	443
Rancho Nuevo	72	100	3,456	405	289
Limón	76	89.5	3,607	422	302
Average		87.8	6,095	713	510
Lowlands					
Loma Final	168	13.1	768	90	64
Cerro Guzmán	296	31.1	3,654	428	306
Carretas	89	15.7	267	32	22
El Mango	151	27.1	1,820	213	152
Average		21.8	1,627	190	136
Average of lowlands and uplands		**54.8**	**3,861**	**452**	**323**

Box 6.3 Taking local knowledge into consideration when selecting tree species for dry forest restoration in central Veracruz, Mexico

A. Suárez and G. Williams-Linera

Tree species selected for tropical forest restoration are often restricted to very few well-known timber species that are seldom native species. Relatively little is known about tree species selection and appropriate techniques for restoring dryland forests in Latin America (Newton, 2008). The standard strategy is to plant mainly exotic timber species for which technical knowledge is readily available, without consideration of the numerous other uses of woody flora, many of which contribute substantially to rural livelihoods. The assessment of species selection for reforestation and restoration programmes is usually based on the experience of technicians working in government agencies as well as on plant availability in official nurseries. This choice overlooks local knowledge and the needs of local populations, and contributes to the failure of reforestation enterprises. Local knowledge turns out to be a useful instrument for increasing the acceptance and interest of local populations in woody species for restoration, as well as for formulating practical management recommendations. Tree species of interest to the local population are well known in the community and may provide a variety of goods and services.

The main objective of this study was to select tree species for tropical dry forest restoration beginning with a local knowledge-based inquiry. The methods included participatory workshops, interviews, and field walks in five rural communities in the municipalities of Paso de Ovejas and Comapa in central Veracruz, Mexico. The communities were located near forest fragments,

Box 6.3 (cont.)

where peasants use forest resources and have a better knowledge of trees. The workshops were conducted as an open group interview. Three questions were posed: Which native trees are useful? Which native trees are scarce? Which native trees are beneficial for wildlife? Cards of three different colours (one for each question) were handed out, and participants wrote down their name and age as well as the local names of all trees in the category. Open discussion was allowed. People who could not write indicated their answers to a research assistant.

A total of five workshops, two focus group meetings, 40 interviews, and 35 field walks were carried out from May 2007 to November 2008. In the workshops, 95 people ranging from 12 to 84 years participated. The 40 people who listed the most names were considered key informants. They were 36 men and four women, all of them over age 39. Key informants were questioned more systematically with semi-structured interviews on the use and management of woody plants and the wild animals associated with them; their help was requested for field walks. During these, we located the trees mentioned during the workshops and collected botanical specimens from them. Key informants were also asked to complement the preliminary lists with more information on the use, management, and ecological characteristics of the habitat in which the trees occur. Information from workshops and field walks was used to calculate three indices of relative importance for each species: the Cultural Importance Index (CII), Scarcity Perception Index (SPI), and Wildlife Importance Perception Index (WIPI).

Participants in the workshops named between 34 and 47 species in each event, for a total of 76 species for the categories useful (CII), scarce (SPI), and important for wildlife (WIPI). Also, informants classified trees as typical of mature forests ('*matas*'), secondary forests ('*acahuales*'), gallery forests, and agroforestry systems (around fields, living fences, and home gardens; these classifications were verified by direct observation). The species belonged to 29 botanical families. Fabaceae had the most species (18), followed by Bignoniaceae and Malvaceae with five each.

All 76 named species were classified as useful (CII), and the most important categories were rural construction, edible, fence posts, and firewood. The species with six uses, the highest number, were *Chloroleucon mangense*, *Leucaena lanceolata*, and *Tabebuia chrysantha*; *Lysiloma acapulcense*, an important fence post species, was in second place. The ten species with the highest CII value comprised 36% of this index; seven of these were legumes from secondary forests (acahuales). The primary use of most of these was as fuel, with the following species preferred: *Acacia cochliacantha*, *Acacia pennatula*, *Diphysa carthagenensis*, and *Leucaena lanceolata.*

Two-thirds of the species were considered scarce (SPI). In this category, five out of the 10 species with the highest CII were included, with *Diphysa carthagenensis*, *Lysiloma acapulcense*, and *Chloroleucon mangense* considered the most scarce. 'Scarce' does not necessarily mean that they are rare; some are relatively common but, owing to severe exploitation, they are insufficient to meet the demands of local populations. For example, farmers said that *Diphysa carthagenensis* has been over-used for firewood and fence posts. Even though the species re-sprouts from cut stems, there were hardly any trees with sufficient wood for exploitation. *Lysiloma acapulcense* was also over-used but had additional problems for natural and even artificial regeneration.

Two-thirds of the total species were considered important for wildlife (WIPI), particularly species of the Moraceae family. In this study, *Ficus cotinifolia* was perceived as the most important species for animal food and habitat. The 17 most important species with potential for TDF restoration activities were selected because they had the highest values within each index (usefulness, scarcity, and wildlife importance) and add up to one-third of the total index value (**Box 6.5**, **Table 2**). Those species can be recommended based on local knowledge, as they should be widely accepted by local people. They will plant them and encourage natural regeneration, since they are all managed in agroforestry systems and provide a variety of useful products and services. Six of them have particular conservation importance because they produce fruits for wildlife consumption. Thus, their use would mean strong support for conservation efforts.

Box 6.3 (cont.)

If only the 17 species with the highest values of each index are recommended, about one-third of all species are covered. This is a convenient number, particularly as the local population is already familiar with successful – though rudimentary – propagation techniques, and will be concerned with the success of the planting. Only two of these (*Cedrela* and *Tabebuia rosea*) are currently available in the region's nurseries. We strongly suggest promoting the other 15 species through systematic collection of seeds, nursery propagation, and use in forest restoration. Activities should include establishment of local plantations, agroforestry systems, and enrichment plantings of secondary vegetation that would be highly valued by landowners.

Box 6.4 Willingness to reforest with native species in rural communities of central Chile

C. Smith-Ramírez, V. Maturana, J.J. Armesto

Native sclerophyllous forests in central Chile are now largely restricted to steep creeks and uplands. The loss of sclerophyllous forest cover and its degradation started more than 300 years ago, primarily owing to the combined impacts of firewood extraction, woodland burning, and livestock grazing (see Chapter 2). Today, a high proportion of rural habitants in central Chile are poor families that live on less than US $400 per month and subsist on highly degraded land with limited or no forest cover. We asked about the willingness of six rural communities to plant native trees, and discuss these results in the light of the new Chilean Forestry Law, which offers monetary incentives to private initiatives to reclaim non-forested land.

We found that 53.9% of the respondents (N = 217 interviews) were interested in planting native trees if the state pays for the full cost. If the landowner pays for the costs, the interest was 47.0%. The willingness to reforest with native species was related mainly to the use of native trees as sources of honey, firewood or because of their ornamental value, and the geographic proximity of the communities to the forests. Gender and economic status of the respondents, presence of native trees on their land, and the identity of the rural community were not significant variables in the analysis. But, only 23% of the 47.0% interested in planting native trees, were prepared to do so on their own land. The majority were willing to plant on nearby hills, riparian habitats and public land, where the reforestation incentives of the new Chilean Forestry Law are not possible to apply. We concluded that even if monetary incentives to plant native trees as part of restoration plans could be offered extensively to rural communities, it is still necessary first to educate rural inhabitants about the values of native tree plantations.

Box 6.5 Traditional knowledge in the drylands of central Mexico: an endangered resource?

R. Aguilar-Santelises and R. F. del Castillo

The vast traditional knowledge of indigenous people in Mexico has been considered endangered (Caballero and Cortés, 2001; Carlson and Maffi, 2003). An important component of this knowledge, accumulated during millennia, is the result of a close contact between the people and their native species. However, starting with the Spanish conquest, this contact began to dwindle, first at a slow pace and recently at a very fast pace (Arredondo *et al*., 1981; Pérez, 2006). Several factors are blamed for this problem, but all of them are related to two phenomena: one social and one biological. The first one is related to acculturation, a gradual process of change, in which traditional culture is supplanted by modern culture (Zent, 2001; Aguilar-Santelises, 2007). The continuous deterioration and eventual local extirpation of many native species and ecosystems comprise the biological problem (Cotler *et al*., 2007).

Box 6.5 (cont.)

One of the best examples in Mexico, in which these two phenomena can be detected, is in the Upper Mixtec region, in the southern Mexican state of Oaxaca. This is a dryland territory with a great diversity of native species, many of which are endemic (García-Mendoza *et al*., 1994), but which also displays one of the highest rates of soil erosion in the world (ca. 200- 280 tonne y^{-1}; Anonymous, 2007). The Mixtecs settled in this region ca. 1400 B.C. and initiated a rich culture with their own language. As an effort to rescue part of the traditional knowledge of native plants in this region and to increase our understanding of the process of acculturation, we recorded the uses of native woody species and compared the distribution of the traditional among the local people in three municipalities of the Upper Mixtec region. Of particular interest is that these municipalities display contrasting proportions of Mixtec-speaking people, formal education levels, and public medical services (Anonymous, 2005). This suggests different degrees of contact with the preponderant culture, allowing a test of the hypothesis that acculturation is a primary factor contributing to local extirpation of traditional knowledge.

We collected information about native plant uses by showing each informant a set of herbarium specimens of 112 species of native species randomly chosen from the locality, and asked for the local name of the plant and uses he or she gave to each of the shown species (**Fig. 1**). We evaluated the importance of each species by the proportion of informants that reported such species to be useful, and the distribution of knowledge according to the native language proficiency and education level of the informants. The recorded uses were classified into eleven categories. All of the species were reported useful by at least a few of the informants. However, the importance of the species to the community was unevenly distributed: a few species were described as highly useful, whereas most of the species were found to be useful only to a minority (**Fig. 2**). *Juniperus flaccida* was the most useful species, as more than 90% of the informants reported this species as useful. *Leucaena*, a legume tree, and six oak (*Quercus* spp.) species followed as the next most useful species (**Table 1**). Native plants were mostly used as livestock food and medicine, fuel (firewood and charcoal), and human food (**Fig. 3**).

Regarding the traditional knowledge among local people, we found that it was strongly rooted in the study area. However, few people were aware of the usefulness of a high proportion of the species shown, whereas most people found useful only a small fraction of those species (**Fig. 4**). Interestingly, the municipality with the lowest evidence of contact with the preponderant culture showed the highest level of traditional knowledge (**Fig. 5**).

This study has shown that traditional knowledge about plant uses in the Upper Mixtec region is very rich. However, it is unevenly distributed both among people and municipalities. Preserving this valuable knowledge requires supporting the dissemination of this knowledge among the population. This could be achieved by the implementation of workshops led by the most knowledgeable people, and the eventual incorporation of this knowledge into formal education.

Figure 1 One of the interviewees examining a botanical specimen of a native species in the Upper Mixtec region, Oaxaca, Mexico.

Box 6.5 (cont.)

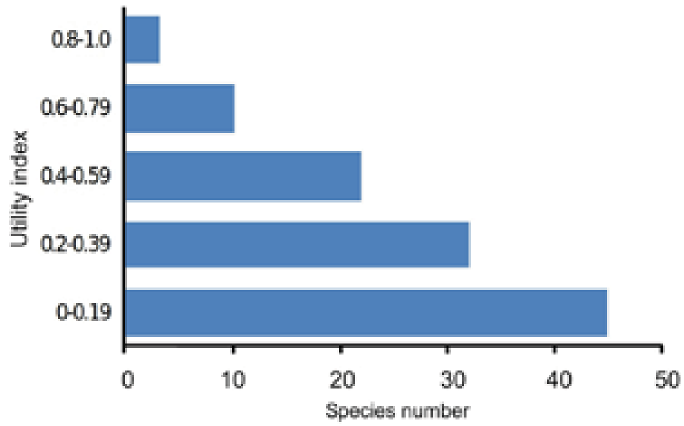


Figure 2 Distribution of the utility index (frequency of informants reporting some use) among the native plants examined by the informants in the Upper Mixtec, Oaxaca, Mexico.

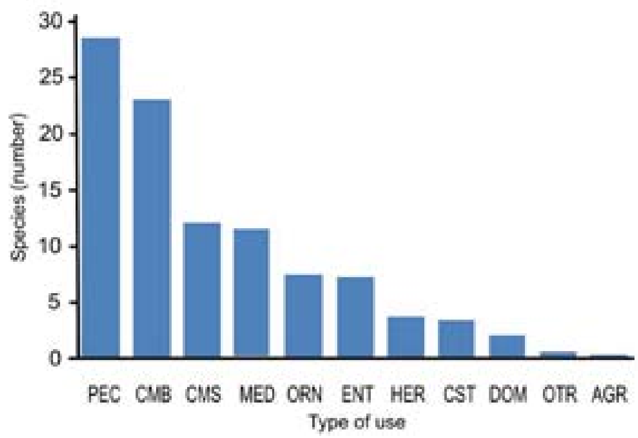


Figure 3 Main types of use of native plants in the Upper Mixtec, Oaxaca, Mexico. PEC = livestock food and medicine; CMB = fuel; CMS = edible; MED = medicine, ORN = ornament; ENT = entertainment; HER = tools; CST = construction; DOM = domestic; OTR = other; AGR = agriculture.

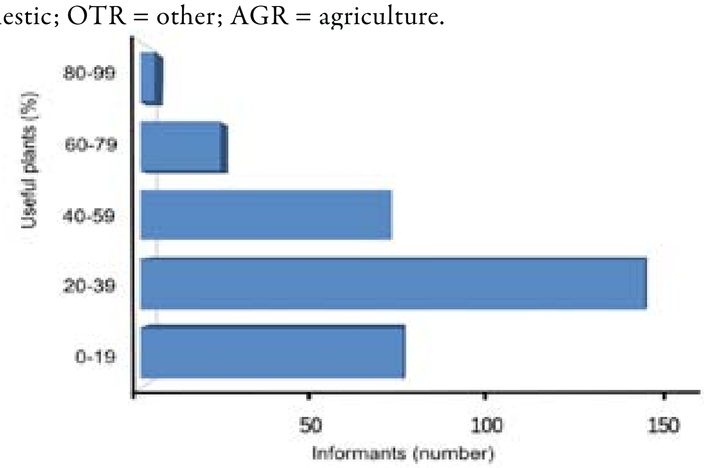
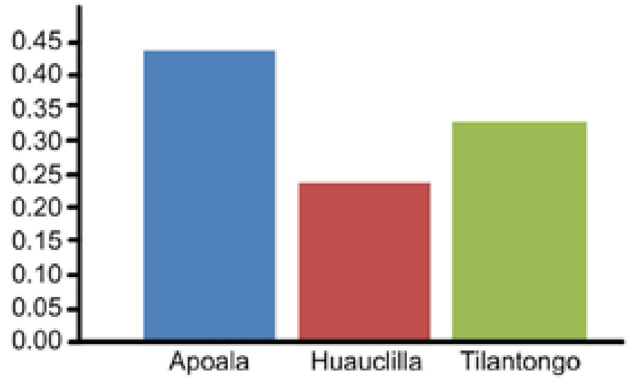


Figure 4 Distribution of the percentage of native plants found to be useful per informant in the Upper Mixtec region, Oaxaca, Mexico.

Box 6.5 (cont.)



	Education level	Language proficiency
Apoala municipality	4.43	2.70
Huauclilla municipality	5.84	0.05
Tilantongo municipality	5.82	1.52

Figure 5 Average Utility Index in three municipalities (Apoala, Huauclilla, and Tilantongo) of the Upper Mixtec region, Oaxaca, Mexico. The table below the graph shows the average number of years of formal education per inhabitant, and their proficiency (0 = do not understand, 3 = fully qualified domain) in Mixtec language in these three municipalities.

Table 1 The most frequently used native forest species in three municipalities of the Upper Mixtec, Oaxaca, Mexico.

Species	Utility index
Juniperus flaccida	0.92
Leucaena diversifolia	0.86
Quercus acutifolia	0.84
Quercus laurina	0.79
Quercus deserticola	0.74
Quercus castanea	0.74
Quercus liebmannii	0.73
Acacia pennatula	0.73
Quercus rugosa	0.72
Acacia farnesiana	0.69
Arctostaphylos pungens	0.65
Tecoma stans	0.63
Litsea glaucescens	0.61

Seasonal dry premontane forest in northwestern Argentina. Photo: L. Malizia

Box 6.6 Assessing the value and commercial potential of non-timber forest products: the CEPFOR project

A.C. Newton

As noted elsewhere in this chapter, many rural communities are strongly dependent on a range of non-timber forest products (NTFPs) to support their livelihoods. While many of these are used for subsistence purposes, in recent years there has been growing interest in the commercialization of NTFP, as an appropriate means of developing forest resources. This reflects a growing recognition of the contribution made by many NTFPs to rural livelihoods, both in terms of supporting subsistence and as a means of generating financial income. At the same time, as harvesting of NTFPs is generally considered to be less damaging to forest resources than timber extraction, the production of NTFPs is widely believed to be relatively compatible with forest conservation (Arnold and Ruiz Pérez, 1998). Thus commercialization of NTFPs potentially offers a means of achieving both conservation and development goals concurrently, by increasing the value of forest resources to local communities.

Recent reviews suggest that approaches to NTFP commercialization have not, however, been universally successful, and that scope for improving rural livelihoods through NTFPs is in doubt (Sheil and Wunder, 2002). In a comprehensive review of NTFP commercialization, Neumann and Hirsch (2000) indicate that sale of NTFPs often tends to provide a low level of income for the poorest sections of communities, rather than providing a method of socioeconomic advancement. The NTFP trade may actually perpetuate poverty rather than alleviate it (Neumann and Hirsch, 2000). Given that NTFPs are highly diverse in terms of their ecological and socioeconomic characteristics, there is a need to define which NTFPs have particular potential for development, and under what conditions their use is likely to make a positive contribution to both human livelihoods and forest conservation. Such information would help reduce the misdirection of donor investments identified by Sheil and Wunder (2002).

This issue was recently addressed by an interdisciplinary research project (CEPFOR), which examined the factors influencing NTFP commercialization in 19 case studies from Mexico and Bolivia, through an intensive programme of participatory research conducted with local communities and other stakeholders. A number of these case studies were located in tropical dry forest areas. Results of the research are presented in detail by Marshall *et al.* (2003, 2006), and outputs of the project are freely available from this website: http://quin.unep-wcmc.org/forest/NTFP/. This includes guidance on appropriate methods for assessing the value of forest resources.

Key findings of the research included the following:

- NTFPs are important in the lives of the rural poor, and incomes vary greatly even between households engaged in the same activity. NTFP activities were found to contribute between 7% and 95% of a household's annual cash income; regularly provide a safety net for the poor to fall back on when other activities – such as subsistence agriculture or cash crops like coffee – fail to deliver as expected; and sometimes provide a stepping stone to a non-poor life, and never lead to an increase in poverty.
- The importance of NTFPs in household livelihood strategies is closely linked to their seasonality and the way they may be combined with other income-generating activities.
- The more months a product can be traded, the more favourably households view the activity. Conversely, households involved in seasonal products are more likely to switch from NTFP activities to other livelihood options, reflecting their desire for a more consistent and year-round source of income.
- In the case of communally-owned resources, improved management of the natural resource and better harvesting practices are common. If land is held privately and the plant can be easily propagated, individuals begin to engage in small-scale domestication. Many of the communities expressed a strong interest in cultivating plant species of high value for NTFP production, highlighting the economic potential and community support for forest restoration efforts.

Acacia pennatula **in a pasture that that has been pollarded for firewood and secondarily for forage, Chiapas, Mexico. Photo: B. Ferguson**

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7 IMPACT OF FOREST FRAGMENTATION AND DEGRADATION ON PATTERNS OF GENETIC VARIATION AND ITS IMPLICATION FOR FOREST RESTORATION

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Introduction

Dry forests are currently the focus of increasing conservation and restoration efforts. This is because one billion people live in dry regions of the world that cover nearly 40% of the Earth's surface. These regions all have in common a reliance on natural resources – including biodiversity, which is declining at a rate unprecedented in recorded history (UNDP, 2004). The objectives of the ReForLan project were to identify and promote approaches for restoration of arid and semi-arid forest ecosystems. The focus of this chapter is to assess the impact of forest loss, fragmentation, and degradation on genetic variability within socioeconomically important tree species of conservation concern, in the context of restoring functional forest landscapes. In addition, the chapter provides recommendations for restoration of dryland forest resources based on an understanding of the processes influencing genetic variation.

Patterns of genetic diversity in plants are the result of current and past evolutionary processes that can be used to guide conservation efforts. Molecular markers can be of great value for investigating the effects of neutral processes such as genetic drift affecting small populations and isolation owing to barriers for gene flow. Such events tend to erode genetic variation in natural populations. Markers may evolve at distinct evolutionary rates and therefore can provide information about processes acting at different temporal scales. Mutations per generation of uniparentally inherited DNA markers such as those of the chloroplast occur at rates of about 10^{-9}, whereas for nuclear microsatellites, mutation rates are between four and six orders of magnitude greater (Provan *et al.*, 1999). While sequences of chloroplast DNA can be used to reconstruct historical genetic patterns, nuclear markers may elucidate the contemporary genetic structure of natural populations. Hence, the combination of the two markers gives the opportunity to understand past and present genetic patterns so as to guide conservation and restoration efforts for the long-term preservation of species.

Our aim was to analyze patterns of within- and between-population genetic variation in species of conservation concern, economic importance, and/or socioeconomic relevance in three areas of Latin America. Different markers were used for genetic analyses of natural populations. These included traditional isozyme methods, and novel molecular analyses such as Single Nucleotide Polymorphism (SNP), nuclear species-specific Simple Sequence Repeats (SSRs) known as microsatellites, and DNA sequences of non-recombinant regions of the chloroplast. Whereas the latter provides a historical signal, the former three mostly reflect contemporary genetic structure.

Mapache (*Procyon lotor*) found in one of the study sites in central Veracuz, Mexico. Photo: C. Alvarez

Open *Austrocedrus chilensis* stands at the forest-steppe ecotone in southern Argentina. Photo : T. Kitzberger

Genetic patterns were analyzed in three geographic areas: northern and southern Argentina and Oaxaca, Mexico (see also **Boxes 7.1** and **7.2**). In these study areas the following species were studied: three *Cedrela* species, *C. balansae*, *C. lilloi*, and *C. saltensis,* from seasonally dry subtropical forests of northern Argentina (Yungas); the dominant tree *Austrocedrus chilensis* (hereafter *Austrocedrus*) of the forest-steppe ecotone of the Patagonian Andes of southern Chile and Argentina; and *Malacomeles denticulata* and *Catopsis berteroniana* from arid and disturbed environments in the central state of Oaxaca, Mexico.

Box 7.1 Genetic hotspots: the quest for preservation of Chile's evolutionary history

M. Henríquez, C. Echeverria, A. Premoli, G. Machuca, C. Souto and P. Quiroga

A biological hotspot is defined as a bio-geographic region with a significant amount of biodiversity that is under threat of destruction owing to human action. The importance of knowing and preserving these hotspots lies in the fact that they continue the evolutionary history of the world and play a role in the biological equilibrium of the biosphere. In order to achieve the goal of conserving a biological hotspot, it is necessary to prioritize threatened areas so that we can identify which are the most valuable spots to preserve. However, since the biodiversity hotspot concept was originated, genetic information has not been considered in their designation.

Owing to its geography, Chile has been called a bio-geographic island. Because of that, Chile provides a unique centre of global biodiversity: the Chilean Winter Rainfall Valdivian forests (Dinerstein *et al*., 1995). During the last 40 years, this biodiversity hotspot has been affected by deforestation and land-use change. To determine which areas are important for biodiversity conservation, we used three large databases on the presence of threatened species within the bio-geographic region: the Catastro y evaluación de recursos vegetacionales nativos de Chile (CONAF *et al*., 1999), a Darwin Initiative Project (Hechenleitner *et al.,* 2005), and data collected by the ECOTONO laboratory of the Universidad del Comahue in Argentina. More specifically, we used data from 26 threatened species to study their distribution patterns, and of these we used genetic information collected from 12 species.

To integrate the data we used ArcGIS software (provided by ESRI). First, we constructed a 5 x 5 km raster map that showed cells where the species are found. The treatment of the cells included an overlay of the locations of the species, in order to construct a richness map showing the areas with more than one species. Then, we generated genetic raster maps consisting of the presence or absence of unique alleles and a heterozygosity scale. These genetic data were obtained through both isoenzyme and DNA techniques, based on the presence of molecular and biological markers. This information, when combined with the species richness map, allowed us to determine genetic hotspots. These hotspots were defined as those areas with presence of at least one species with unique alleles, at least one species with high heterozygosity and species richness at least equal to 3.

Our results revealed the presence of high richness areas along the coastal range in Chile and the Andean range both in Chile and Argentina (**Fig. 1**). Most richness areas were located in the coastal range in the Maule and Bio-bio Regions. The species that occur in these areas are highly threatened as they are located in one of the areas with the highest rates of deforestation and land-use change in Chile (Echeverria *et al*., 2006). Populations with unique alleles were distributed in four diagonal lines from the coastal range in Chile to the Andean range on the Argentinean side (**Fig. 1**). Four areas share populations of two species with unique alleles: one near Curacautin (Araucanía Region, Chile), one in Valdivia (Los Ríos Region, Chile), one to the north of Bariloche (Neuquén Province, Argentina) and one near Esquel (Chubut Province, Argentina). Populations with high heterozygosity are mainly concentrated in Argentina, from Neuquén to the Chubut Provinces **(Fig. 1**); only a few populations are found across the Chilean side both in the coast and in the Andes.

Box 7.1 (cont.)

Five genetic hotspots were identified only in Chile in the zones of Cauquenes, Bullileo, Curacautín, Valdivia and Puyehue (**Fig. 1**). Of these hotspots, four are not protected by the Chilean state; only Puyehue is completely located within national parks. It is important to note that of the five hotspots identified, three of them (Curacautin, Valdivia and Puyehue) possess populations of two species with high heterozygosity (**Fig. 1**).

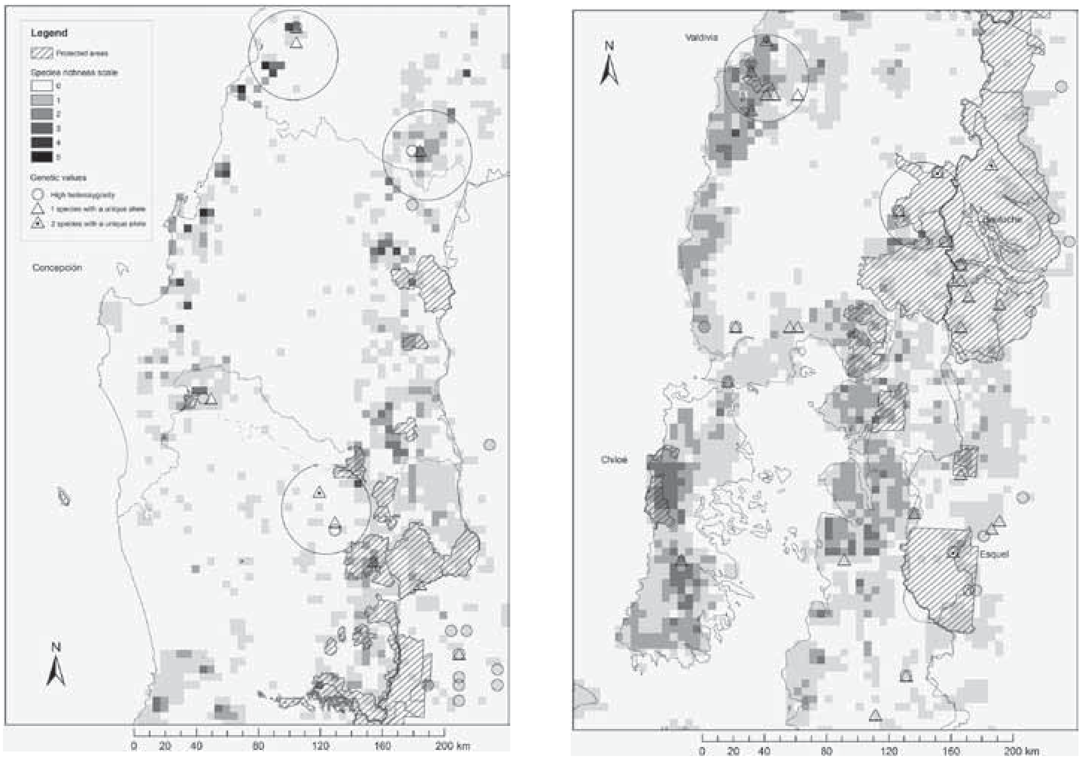

Figure 1a, b Distribution of richness of threatened tree species, zones of high genetic diversity and genetic hotspots. Large circles represent genetic hotspots determined by the presence of unique alleles, high heterozygosity (level 4) and at least three threatened species occurring simultaneously in the area. Figure 1a illustrates central Chile and Argentina, and Figure 1b illustrates an area immediately to the south.

The genetic hotspots identified in the present study are candidate areas for protection because of the valuable genetic information and high biodiversity value contained there. These genetic hotspots can be considered as biogeographic regions with a significant number of genetic resources threatened by human actions and under risk. The most important issue about these genetic hotspots is that they are the basis for species adaptive potential, which will determine their ability to survive environmental change such as climate change. Because of the heterogeneous nature of the study landscapes, the management of the area needs to be undertaken at different scales. The next step has to be the development of multi-scale management policies to ensure the preservation of those habitats that hold species with different phylogenies, and to protect the populations not strongly affected by human beings, because they retain unaltered phylogeographic characteristics. This kind of management would allow us to understand how ecological processes change according to scale.

Box 7.2 Testing forest connectivity using genetic distance: spider monkeys and dry forest restoration, Nicaragua

S. Hagell, S. Otterstrom, C. Chambers

Wildlife across Central America is under immediate threat from forest loss and fragmentation. These threats are particularly evident in the critically endangered Central American Dry Forest (WWF), in which it is estimated that less than 5% of native forest remains intact or protected (Miles *et al*., 2006). We are conducting research to understand how wildlife, particularly endangered or threatened species, negotiate these human-dominated landscapes in south-western Nicaragua. As a part of ongoing forest restoration initiatives, Paso Pacifico is working with wildlife biologists at Northern Arizona University to study landscape genetic patterns in the Central American spider monkey (*Ateles geoffroyi*). The spider monkey is a highly charismatic arboreal mammal that is also a key seed disperser for native trees (Pancheco and Simonetti, 2000). Our goal is to preserve the spider monkey population in this landscape by restoring dispersal corridors between forest fragments. However, because these animals are rare and elusive, there is limited behavioural information as to the dispersing animals' area in the current landscape. For this project, we are using non-invasively collected genetic samples from faecal material to test our hypotheses as to how spider monkeys may be able to use secondary and non-forest matrices for dispersal. More specifically, we are comparing mean genetic distances between social groups to the spatial distance calculated from alternative landscape resistance models. These models are based on the cost of travel through mature forest, regenerating or secondary forest, and non-forest matrices. This is a relatively new approach to conduct spatial genetic analyses, because genetic data is used as a means to measure dispersal cost and test multiple hypotheses of landscape connectivity (as in Cushman *et al*., 2006; Epps *et al*., 2007). Furthermore, our analyses provide estimates of resistance for each landscape feature in the model, data that can be used directly to build a predictive model for the whole landscape. As an example, the 'best' models of landscape resistance can be combined with new circuit theoretic tools to identify 'pinch points' that are critical for connectivity (McRae *et al*., 2008). In this way, the patterns of genetic diversity in this landscape will reveal pathways and barriers to dispersal, and indicate how forest management can be used to preserve this species.

Seasonally dry subtropical forests of northern Argentina

Cedrela species are among the most valuable hardwoods of Argentina. Consequently, they have been heavily logged within the study area. Since recruitment of these species is disturbance-dependent and they display high growth rates, they are ideal for forest restoration and stand enrichment purposes (see also Chapter 6).

In subtropical seasonally dry forests of Argentina, the distribution of dominant woody *Cedrela* species follows elevation gradients. *C. balansae* occurs in piedmont forest between 300 and 800 m in elevation attaining its southern limit at 24°30'S. *C. lilloi* is found in montane forest between 500 and 1350 m reaching the southern-most distributional range of the genus at 28°15' S latitude. These two forest types, piedmont and montane, are the most economically important (Brown and Pacheco, 2006). More recently a third species, *C. saltensis*, has been identified. Apparently, the latter has a restricted range (<1000 km^2) (Malizia *et al*., 2006) and occurs in an area of sympatry between the other two species at elevations of between 700 and 1100 m a.s.l. (Grau *et al*., 2006), reaching its southern limit at 24° 40' S latitude.

We used molecular marker techniques to test the hypothesis that *C. saltensis* is a species of hybrid origin between *C. balansae* and *C. lilloi*. The main purpose was to use information on patterns of genetic differentiation between the two parent species and putative hybrids to define appropriate sources of germplasm to guide future restoration and conservation actions.

Interspecific hybridization is a common phenomenon in plants. The frequency of hybridization in trees is particularly related to their longevity and various reproductive systems that allow interespecific gene flow. However, hybrid formation will depend upon the genetic compatibility between species, the flowering phenology, and their degree of range overlap. In addition, favourable micro-site conditions, known as 'hybrid habitats' (Anderson, 1948) may facilitate the establishment and survival of hybrid progeny. These are usually related to disturbed sites and/or locations where pure plants have relatively reduced competitive potential and fitness.

To study the possible hybrid origin of *Cedrela saltensis* from *C. balansae* and *C. lilloi* we sampled pure species and potential hybrids inhabiting seasonally dry forests of northern Argentina. A genetic study of three species of *Cedrela* was conducted on populations located between 23º 5' and 24º 30'S latitude near to Calilegua National Park. We collected fresh leaf tissue from natural populations of each species (**Table 7.1**) which were analyzed using 12 isozyme loci. These were Glycerate 2 dehydrogenase (G2d), Isocitrate dehydrogenase (Idh1, Idh2), Malic enzyme (Me1, Me2), Phosphoglucoisomerase (Pgi1, Pgi2, Pgi3), Peroxidase (Per1, Per2, Per3), and Shikimate dehydrogenase (Skdh). We calculated standard population genetic parameters. These included estimates of diversity and inbreeding at the population and species level, respectively. We estimated genetic distance metrics and ran multivariate cluster analyses to portray genetic relationships among species. Heterogeneity in allelic frequencies across populations and species was analyzed using chi-square tests.

Results showed differences among the three analyzed *Cedrela* species in allelic frequencies and diagnostic alleles, i.e. those present exclusively in just one species, were found in all studied species. *Cedrela balansae* had seven, *C. lilloi* had one, and *C. saltensis* had six diagnostic alleles, respectively (**Table 7.2**). All three species were genetically diverse although *C. saltensis* had greater heterozygosity (He = 0.330) and mean effective number of alleles (Ne = 1.63) than putative parent species *C. balansae* (He = 0.229 and Ne = 1.39) and *C. lilloi* (He = 0.276 and Ne = 1.48). In addition, estimates of within-population inbreeding yielded Fis values of 0.135 (CI 0.016 – 0.211) for *C. balansae*, 0.308 (CI 0.050 – 0.640) for *C. lilloi*, and 0.275 (CI –0.061 – 0.642) for *C. saltensis*. The three species were significantly different in their allozymic profiles for 10 out of 11 tests. Genetic distance indices and multivariate cluster analysis by means of Nei's genetic distance (1978) showed that *C. saltensis* is more similar to *C. lilloi* than to *C. balansae* (**Fig. 7.1**).

Table 7.1 Sampled populations of *Cedrela* from northern Argentina.

Species	*Population*	*N*	*Sº Latitude*			*Wº Longitude*			*Elevation m a.s.l*
C. balansae	R34CB	27	24	17	46.1	64	54	33.7	730
C. balansae	PNCCB	3	23	46	54.7	64	48	54.6	535
C. balansae	SSCB	22	23	40	28	64	33	49.1	380
C. balansae	LNCB	1	23	7	1.9	64	40	34.7	800
C. balansae	CICCB	32	23	7	39.4	64	27	56	495
C. lilloi	SSJCL	15	24	9	48.4	65	18	50.6	1390
C. lilloi	PSCL	4	24	30	7.7	65	18	43.5	1400
C. lilloi	FTCL	5	23	0.5	11.1	64	51	30.9	1710
C. saltensis	PNCCS	15	23	41	31.9	64	52	44	1450
C. saltensis	CRCS	3	23	5	38.4	64	44	30.4	1000
C. saltensis	ACCS	5	23	5	14	64	47	28.6	1200
C. saltensis	LNCS	5	23	6	59.9	64	40	52.1	830

N: number of sampled individuals

Soil erosion in dry forests in Oaxaca, Mexico. Photo: R.F. del Castillo

Ripening fruit of *Cochlospermum vitifolium* in Central Veracruz, Mexico. Photo: C. Alvarez

Table 7.2 Allele frequencies for isozyme polymorphic loci in *Cedrela* species from subtropical forests of northern Argentina. Diagnostic alleles, i.e. those present in just one species, are depicted in **bold** and null alleles, i.e. absence of protein product, are shown in italics. (*significant allele frequency heterogeneity across taxa).

Locus	*C. balansae*	*C. lilloi*	*C. saltensis*
G2d			
1			**0.306**
2			**0.694**
3	**0.075**		
4	**0.925**		
5		*1.000*	
Idh1			
1*	0.021	0.446	
2*	0.031	0.464	
3		**0.089**	
4			**0.346**
5			**0.615**
6			**0.038**
7	**0.031**		
8	**0.876**		
9	**0.041**		
Idh2			
1	0.969	0.962	1.000
2	0.031	0.038	
Me1			
1		**0.071**	
2	0.901	0.875	0.925
3	0.099	0.054	0.075
Me2			
1	0.862	0.760	0.929
2	0.138	0.240	0.071
Pgi1			
1*	0.138	0.690	0.768
2*	0.862	0.310	0.232
Pgi2			
1	**0.021**		
2	0.155	0.086	0.018
3	0.454	0.603	0.446
4	0.325	0.224	0.018
5	0.041	0.086	0.179
6	0.005		0.036
7			**0.304**
Pgi3			
1		0.042	0.053
2*		0.958	0.816
3*	0.887		0.132
4	**0.112**		
Per1			
1	0.051	0.172	
2*	0.020	0.724	0.741
3*	0.929	0.103	0.259
Per2			
1	1.000	1.000	1.000
Per3			
1	0.116	0.065	0.261
2	0.820	0.870	0.739
3	0.064	0.065	
Skdh			
1	0.022	0.083	0.204
2	0.789	0.854	0.593
3	0.172	0.042	0.204
4	0.017	0.021	

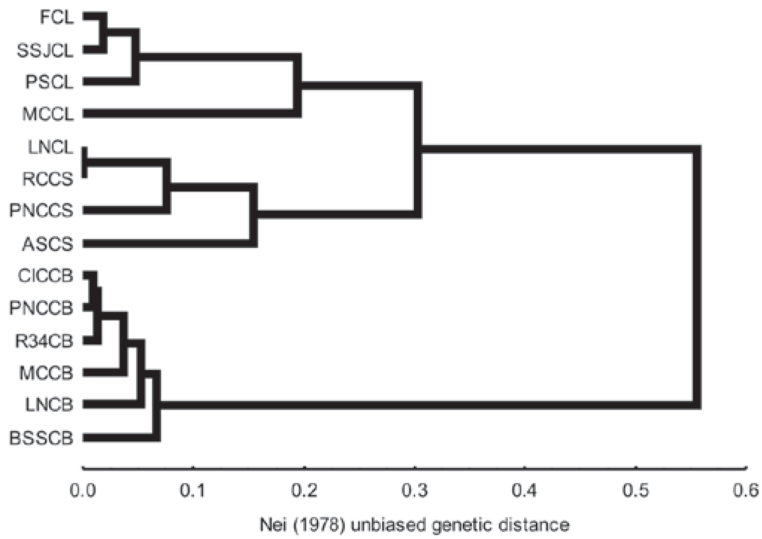

Figure 7.1 UPGMA multivariate cluster analysis using the Nei's (1978) genetic distance for 14 populations analyzed of *Cedrela.* Population names are indicated in Table 7.1. Populations MCCB and MCCL are cultivated individuals from laboratory-grown shoots that were collected in northern Argentina.

Genetic diversity was high in the three *Cedrela* species. In contrast, elevated inbreeding coefficients indicated by Fis in all species probably reflect biparental inbreeding as a result of localized pollen movement by insects. As expected, the putative hybrid *C. saltensis* presented high heterozygosity in comparison with *C. balansae* and *C. lilloi*. Nevertheless, this result *per se* would not be evidence of hybridization. *C. saltensis* was relatively more similar to *C. lilloi*, as shown by a smaller genetic distance between them (**Fig. 7.1**). In contrast, *C. saltensis* is more similar morphologically to *C. balansae* than to *C. lilloi* (Zapater *et al.,* 2004). On the other hand, the presence of diagnostic alleles in *C. saltensis* suggests that this species is taxonomically distinct. A previous molecular phylogenetic analysis suggests that the possible ancestral taxa of *C. saltensis* would have had a separate evolutionary history (Muellner *et al.*, 2009). While more studies are needed to understand the phylogenetic relationships among these taxa, our results strongly suggest that *Cedrela saltensis* is not the result of recent and/or recurrent hybridization between *C. balansae* and *C. lilloi*. Therefore *Cedrela saltensis* needs to be treated as a separate taxonomic entity.

Information on tree species diversity and structure of seasonally dry subtropical Yungas is limited (Malizia *et al.*, 2006; Brown and Pacheco, 2006). Data on distribution patterns of genetic diversity on species of economic and/or conservation concern is also scarce (Quiroga and Premoli, 2007). This information is needed given that selective logging of *Cedrela* species is eroding genetic diversity because the better genotypes with optimal characteristics are continuously being removed. Such processes have been documented in closely related species in Central America (Gillies *et al.,* 1997; 1999). However, natural populations of *C. lilloi* are protected in many different geographic areas, e.g. within National Parks. In contrast, piedmont mature *C. balansae* forests no longer exist in the wild and only young stands occur in marginally protected areas such as nearby National and Provincial Parks. On the other hand, the distribution range of *C. saltensis* has been delimited only preliminarily (Malizia *et al.,* 2006). According to ecological niche models developed for *C. lilloi* and *C. balansae*, the potential area of distribution of *C. saltensis* in northwestern Argentina is about 600 km^2 (Malizia *et al*. 2006). *C. saltensis* stands might occur from southern Bolivia (Tariquía Reserve), across Salta (Baritú National Park) and Jujuy (Calilegua National Park) provinces. Further studies on the distribution range and genetic patterns of this species are needed to fully evaluate its conservation status.

Arid central Oaxaca, Mexico

Malacomeles denticulata (formerly *Amelanchier denticulata*), is a shrub member of the Rosaceae family. Along its distribution from central México to Guatemala, *M. denticulata* grows mainly in disturbed shrublands and pine-oak forests (Rzedowski and Calderon, 2005). As a tolerant species of such arid and degraded environments, it is considered ecologically important for recovering degraded areas.

One of the most recently developed molecular tools to study genetic variation are single nucleotide polymorphisms (SNPs).These are the result of either transition or transversion mutation events. SNPs are single base pair positions in the genome of two or more individuals, at which different alternative sequences or alleles exist in populations (Weising *et al*., 2005). Once SNPs have been determined through sequencing, they can be identified relatively rapidly in any individual of the studied population using real-time polymerase chain reaction (RT-PCR) equipment. RT-PCR monitors in real time the progress of the PCR as it occurs. Data are collected throughout the PCR process rather than at the end of the PCR. Homogeneous detection of PCR products can be obtained by using double stranded DNA binding dyes or fluorogenic probes. Quantification of DNA or RNA can be more precise and reproducible because it relies on threshold cycle (CT) values determined during the exponential amplification phase rather than at the endpoint.

The main objective of this research was to develop and apply a novel DNA marker to access the impact of forest lost and degradation in the genetic variability of *Malacomeles denticulata* in Oaxaca, Mexico where degradation of dry forest is extreme. Leaves were collected for genetic analyses from eight localities of different kinds of habitats, namely tropical dry forest, pine forest, oak pine forest, and chaparral. Parameters of within-population genetic diversity and among-population divergence (Fst) were calculated (see also **Box 7.3**).

Box 7.3 Genetic variability in populations of *Amelanchier denticulata*

J. Ramírez Luis, R.F. del Castillo, E. Cruz Cruz

Amelanchier denticulata is a shrub in the Rosaceae family, which can grow in a wide variety of habitats, including severely degraded and eroded areas. It is a valuable plant as fodder for goats and sheep, and displays a large phenotypic variation among populations. We studied the nature of such variation by means of a common garden experiment including four populations, using a half-sib design in order to divide the total phenotypic variance into its components for three characters: leaf area, leaf indentation, and relative growth rate. Leaf area was the most variable character and had the highest genetic variance, followed by leaf indentation. Relative growth rate displayed little or no detectable genetic variance in the studied populations. Narrow sense heritabilities were also calculated. We found two populations with little or undetectable genetic variation and two genetically variable populations, in particular the San Pablo Huitzo population, which could be a good candidate for breeding and conservation programmes. Finally, we detected evidence of genetic differentiation among the four studied populations, in particular for leaf area and leaf indentation but not for relative growth rate. Different habitat selection regimes may explain the genetic differentiation encountered among populations for leaf area and indentation, whereas natural selection may be more intense in all the studied habitats for relative growth rate.

Stands of *Austrocedrus chilensis* forest associated with rocky outcrops at the forest-steppe ecotone in southern Argentina, Nahuel Huapi, Argentina. Photo: J. Birch

Positive results were obtained with the SNP design assay for *M. denticulata,* which yielded recognizable homozygote and heterozygote genotypes (**Fig. 7.2**). However, heterozygous individuals were only detected in the population of Santo Domingo Yanhuitlan, which was the most degraded locality consisting of denuded soil and gullies. An average Fst = 0.198 provides evidence of genetic differentiation among the eight studied populations. The average negative within-population inbreeding (Fis = –0.282) can be explained by the excess of heterozygosity observed in Santo Domingo Yanhuitlán (observed and expected frequency of heterozygote genotypes were 0.44 and 0.34, respectively).

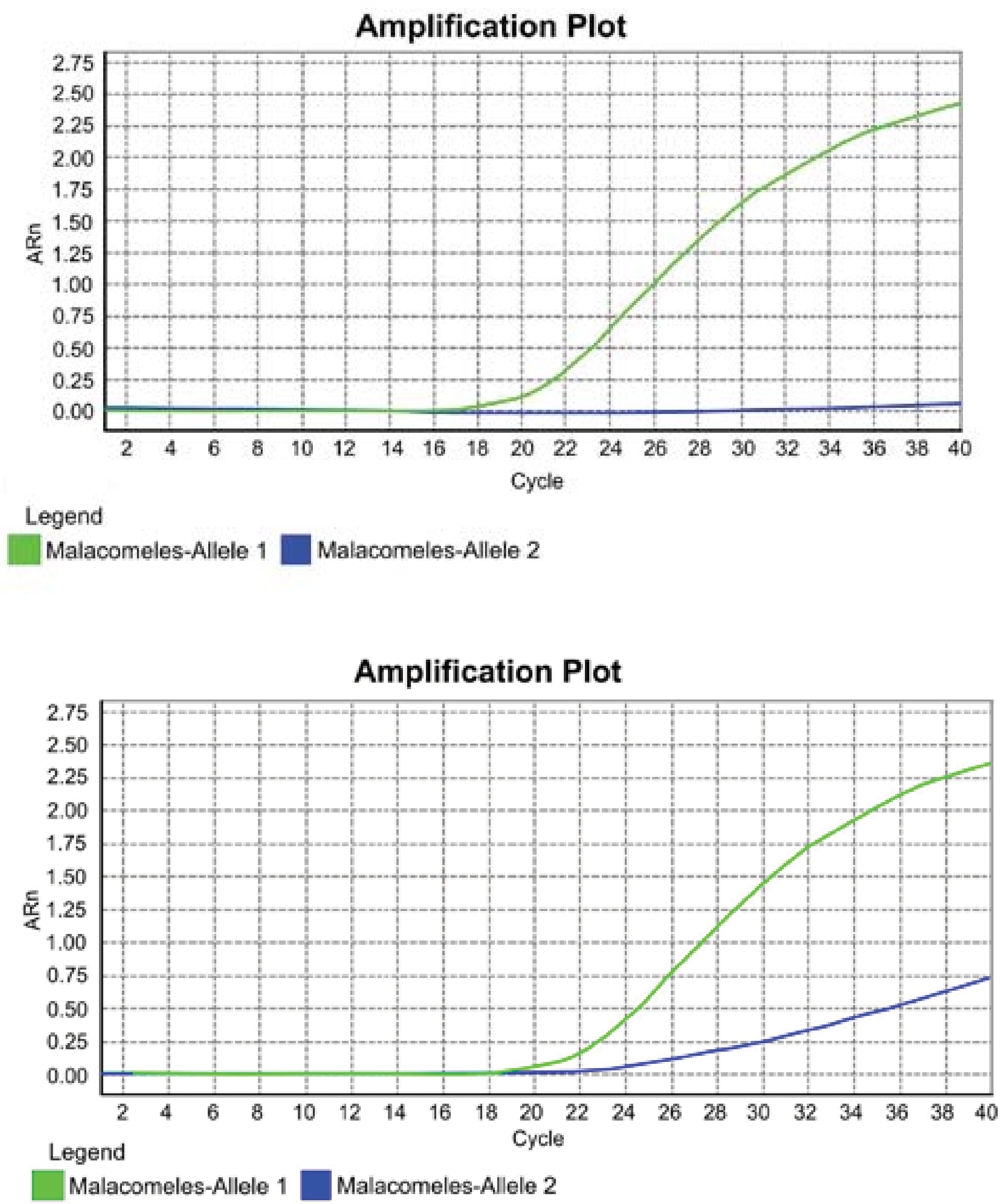

Figure 7.2 Amplification plot of (a) a homozygous individuals and (b) a heterozygous individuals of *Malacomeles denticulata* obtained with a SNP molecular marker.

The population of Santo Domingo Yanhuitlan includes the only heterozygous individuals found in the eight studied populations of *M. denticula*. As a result, it is extremely important to protect this unique genotype. The locality of Yanhuitlan was the most fragmented and degraded of all, so heterosis is probably related to survival mechanisms in *M. denticulata* in this degraded area.

Catopsis berteroniana (Schult and Schult., f) Mez is an epiphyte bromeliad of the Tillandsioideae subfamily. *Catopsis berteroniana* was classified in the Norma Oficial Mexicana 059 (NOM–059-ECOL–2001) as a species that requires special protection. In Oaxaca, México, it is commonly sold in the local markets as an ornamental plant for several religious festivities. Plants are collected by settlers in the localities of El Cerezal and Reynoso in the Santa Catarina Ixtepeji County where the vegetation is a dry fragmented shrubland and disturbed oak forest, respectively. Using satellite images from 1979 to 2005, the rate of deforestation in this region was calculated as 9.1% per decade. Forest remains in fragments that are becoming smaller (**Fig. 7.3**). Owing to forest reduction and fragmentation we expected to find impoverished genetic diversity in these localities.

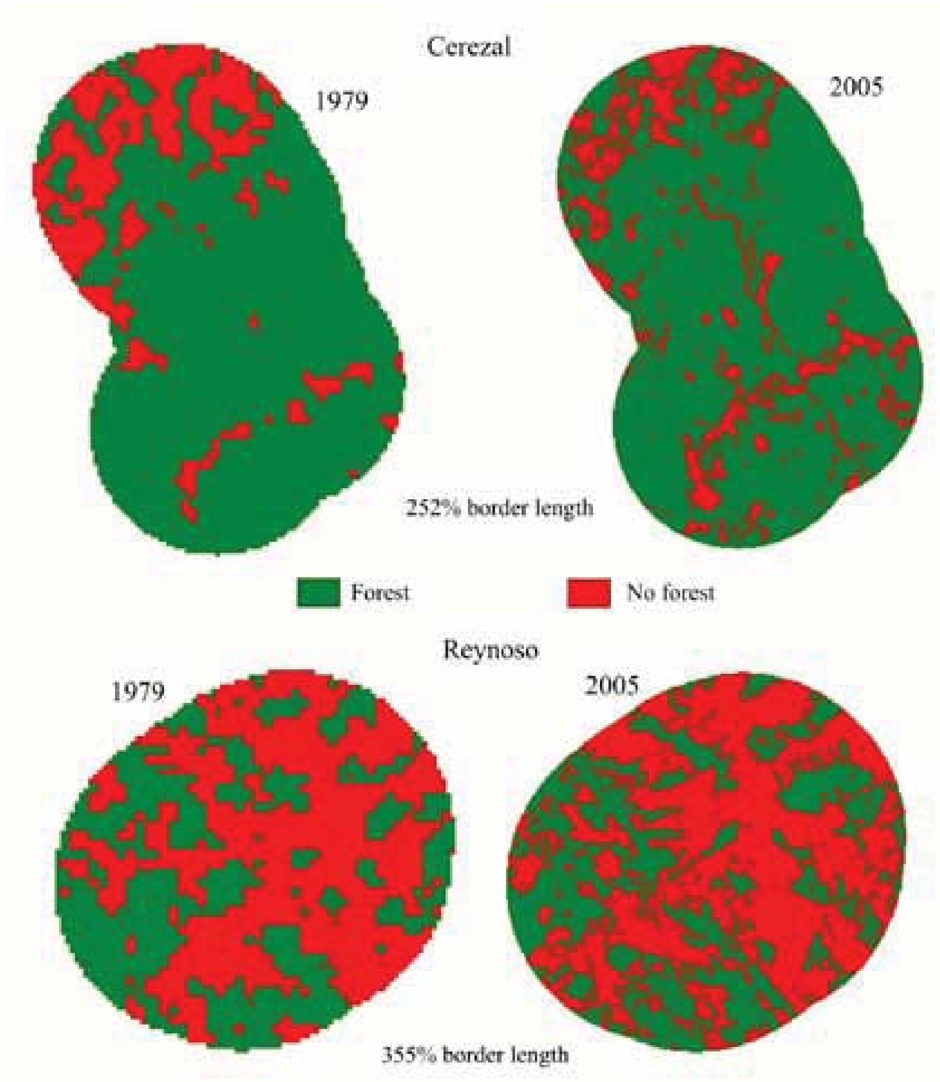

Figure 7.3 Vegetation maps of the localities of El Cerezal and Reynoso, Oaxaca, México, where *Catopsis berteroniana* was studied. Sensor MSS for 1979 maps and Sensor SPOT 5 for 2005 maps were used.

We tested recent bottlenecks on this bromeliad based on patterns of genetic diversity using allozymes. Nine out of the ten analyzed loci were polymorphic. The mean number of alleles per locus was 3.4 and mean observed and expected heterozygosities were 0.402 and 0.292 in El Cerezal and Reynoso, respectively. We found an average within-population inbreeding Fis = 0.256 and a degree of among-population divergence Fst = –0.021 (**Table 7.3**).

Contrary to our expectations, *C. berteroniana* displayed one of the highest levels of genetic diversity recorded in the family. We did not find evidence of recent bottlenecks despite the high levels of forest fragmentation and population reduction (**Fig. 7.4**). In addition, populations in El Cerezal and Reynoso were not genetically differentiated. A demographic study in progress revealed that population size is increasing. Although the total forest area is being severely reduced, the perimeter of the forest fragments has increased, probably favouring gene flow and establishment, as this species appears to grow better on the forest edges. This result shows that some plant species may benefit to a certain extent from forest fragmentation. Nevertheless, increasing fragmentation will eventually reduce fragment edges and thus decrease the available habitat for this species.

Table 7.3 Number of alleles per locus (*A*), number of alleles per polymorphic locus (*Ap*), allelic richness (*Rt*), observed (Ho) and expected heterozygosity (He) found in *Catopsis berteroniana* in the localities of El Cerezal and Reynoso, Santa Catarina Ixtepeji, Ixtlán, Oaxaca, México.

Locality	*A*	*Ap*	*Rt Ho He*
El Cerezal	3.100	3.333	3.079 0.279 0.387
Reynoso	3.333	3.560	3.267 0.305 0.416

F values: Fit = 0.271
Fis = 0.256
Fst = – 0.021

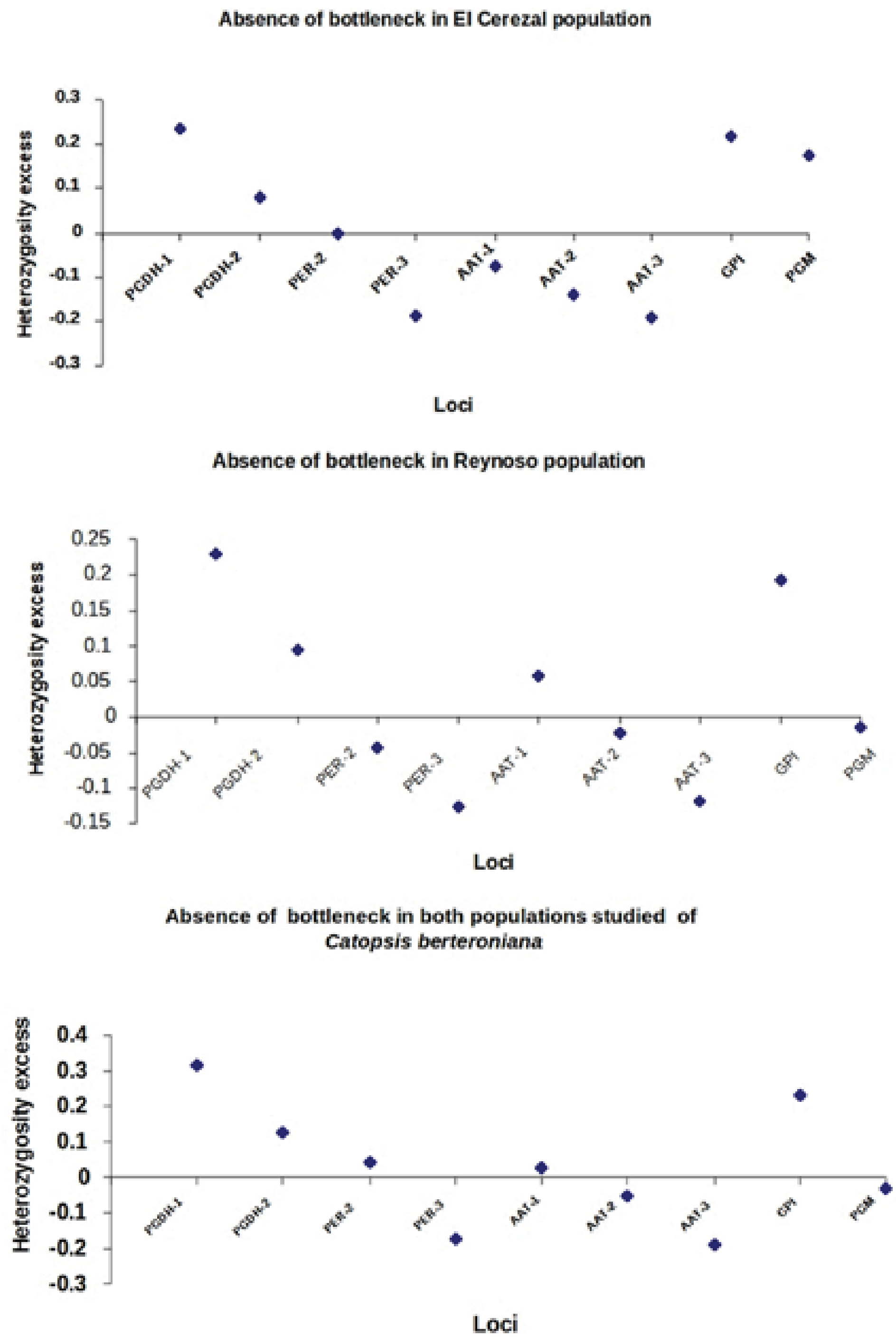

Figure 7.4 Magnitude of heterozygosity excess (He – Heq) observed at each of nine polymorphic loci in the non-bottlenecked population of El Cerezal (a), Reynoso (b) and in both populations studied of *Catopsis berteroniana* (c). The horizontal dashed line represents the heterozygosity excess expected in an equilibrium population with loci evolving under the stepwise model of mutation. Points above and below the X axis line represent loci with a heterozygosity and deficiency, respectively. (Software used: Bottleneck 1.2.02 of the model of Luikart and Cornuett, 1998).
He = Hardy-Weinberg equilibrium heterozygosity.
Heq = Expected heterozygosity in non-bottlenecked populations that are near mutation-drift equilibrium.

Dry temperate forests of southern Argentina

We conducted genetic surveys on *Austrocedrus chilensis*, the dominant tree species of dry warm-temperate forests of Patagonia, Argentina. It is a conifer within the Cupressaceae family that inhabits the forest-steppe ecotone along steep precipitation gradients of more than 1000 mm. It grows in Argentina between 37° and 42° S latitude and from the V to X Region in Chile. *Austrocedrus chilensis* is a timber species of high economic value and international conservation concern. It is listed by the International Union for the Conservation of Nature as Vulnerable (VU A2c; B2ab (iii)). Main threats affecting *Austrocedrus* are herbivory and fire (Hechenleitner *et al*., 2005). The main research objective was to investigate if the gene pool of *Austrocedrus* is geographically structured, and to identify areas with high genetic diversity and/or unique variants. Those areas can be considered to be of high evolutionary potential and/or containing evolutionary novelties that should deserve conservation actions of such valuable and dominant trees. Also, a significant genetic structure along its distribution may guide germplasm collection for restoration actions to be undertaken in degraded habitats. Studies were focused on the understanding of underlying historical and contemporary processes shaping genetic patterns. We combined molecular information from three genetic markers: slowly-evolving DNA sequences of non-recombinant regions of the chloroplast, nuclear isozyme loci with moderate polymorphism, and hypervariable species-specific nuclear markers as microsatellites.

Plastid regions in Austrocedrus

To investigate the historical patterns of genetic diversity and gene flow, we optimized sequences of non-coding regions of mitochondria and chloroplast DNA in an attempt to perform phylogeographic analyses on *Austrocedrus*. Seven different DNA non-coding regions of both, the mitochondria and the chloroplast of *Austrocedrus* were sequenced. These yielded no polymorphism along the *Austrocedrus* range, which impeded phylogeographic analysis. This low mutation rate of the plastid DNA would allow phylogenetic reconstructions within the family, extending to the origin of *Austrocedrus* as a genus. Furthermore, these sequences blasted in NCBI, show affinities higher than 84% with other Cupressaceae such as *Cryptomeria japonica* or *Chamaecyparis sp.*

Isozyme variation in Austrocedrus

A study of the restoration genetics of dry forests of northern Argentina was performed on the dominant tree *Austrocedrus chilensis*. The study was conducted in the forest-steppe ecotone on the eastern slopes of the Patagonian Andes, Argentina, between 37° and 42°S. We sampled a total of 1853 individuals in 67 populations along three regions covering the entire latitudinal range of *Austrocedrus* in Argentina represented as north (N), center (C), and south (S). All sampled locations are moderately disturbed, in terms of logging and grazing, such disturbance being less intense in the central area within Nahuel Huapi National Park. Previous studies have shown that fire is a major driver of ecological constraints in studied locations (Kitzberger, 2003). Regions not sampled included Chilean populations where the climatic and disturbance history of the species are probably older and are highly different to those in Argentina.

To identify the broad-scale trends in genetic differentiation throughout the species' range, the populations were combined into three regions (north, centre, south), according to their geographical proximity and environmental envelope. The north region consists of 23 populations located between 37–39°S latitude characterized by dry climatic conditions and scarce vegetation. The centre region includes 25 populations from 40–41° S where the west-east natural fragmentation gradient is more evident. The south region was based on grouping 19 populations located between 41–43° S where the size of forest patches increases and the mean tree age decreases (K. Heinemann, UNCOMA, unpublished data).

A total of 12 allozyme loci were scored from 1,853 trees distributed over 67 populations. Resolved loci were Glycerate 2 dehydrogenase (G2D), Isocitrate dehydrogenase (Idh), Malate dehydrogenase (Mdh1, Mdh2), Malic enzyme (Me1, Me2), cathodal Peroxidase (Percat), Phosphoglucoisomerase (Pgi1, Pgi2), 6-Phosphogluco dehydrogenase (6Pgd1, 6Pgd2), and Shikimate dehydrogenase (Skdh). All analyzed loci were polymorphic *sensu stricto*, with 3–5 alleles per locus. Genetic diversity metrics of *Austrocedrus* populations show a significant decrease with increasing latitude. Northern populations yielded higher effective number of alleles, total genetic diversity, and allelic richness than southern ones ($F(1,65)$, $p < 0.05$ all tests) **(Fig. 7.5)**.

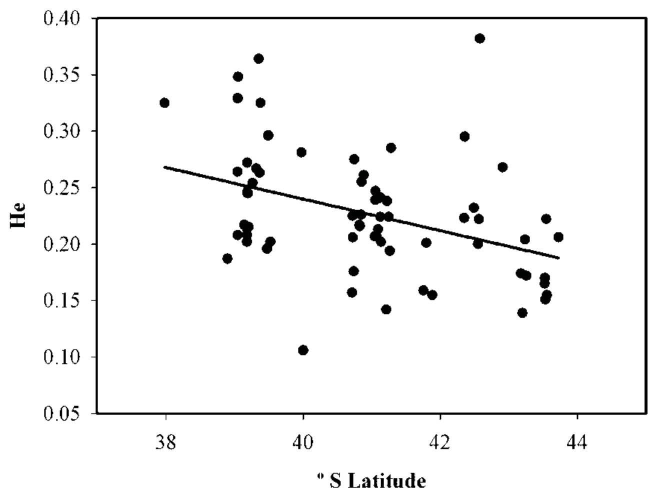

Figure 7.5 Correlation between population genetic diversity (He) and latitude in 67 sampled populations of *Austrocedrus chilensis* in Patagonia Argentina.

Inbreeding within northern and eastern populations of the centre of the range was high, positive, and statistically significantly different from zero (Fis > 0.14). Southern and western populations of the centre of the distribution displayed low levels of inbreeding that do not differ from zero (Fis < 0.06). The degree of genetic divergence between populations was low (Fst <0.16) and similar for the different areas.

Genetic studies performed on *Austrocedrus* show that along its range, populations vary in their rates of diversity and inbreeding. These genetic patterns can be explained in terms of the disturbance history of Patagonia. We hypothesize that fragmented, smaller, and relatively isolated northern populations with higher inbreeding are the result of a long history of human disturbance by fire. In contrast, towards the south, fire history and human impact are more recent and *Austrocedrus* populations consist of more continuous forest stands with lower inbreeding. In addition, the combined effects of human activity and historical factors such as the last glaciations in Patagonia explain the reduced genetic diversity recorded towards the south. This relates to the fact that *Austrocedrus* is a cold-intolerant plant. As a result, it is hypothesized that during cold periods *Austrocedrus* remained at northern warmer latitudes. Therefore, lower genetic diversity found in the south is the result of founder effects suffered during postglacial long-distance dispersal as suggested by the pollen record (Whitlock *et al.*, 2006).

Microsatellite variation in Austrocedrus

In an attempt to combine historical and contemporary signals detected with the previous two molecular markers, nine micro-satellite markers (eight di-, and one trinucleotide bases) were isolated by ATG Genetics, Canada, and characterized for *Austrocedrus chilensis*. Four markers were optimized and scored with confidence, which were tested in 398 individual samples from 43 locations. Study areas included a subsample of eastern populations from southern Argentina used in the isozyme study and seven locations from Chile. The number of scored alleles ranged between 4 and 42 for the analyzed markers. Mean observed and expected heterozygosities were 0.732 and 0.684, respectively. Populations were grouped according to their location in five regions: coastal and Andean populations from Chile and in Argentina represented as north (N), centre (C), and south (S). Results show a reduction in polymorphism towards the coast in Chile and the south of the species' range in Argentina (**Figs. 7.6** and **7.7**). These results suggest that eastern populations have derived from most variable northern Andean sources by long-distance dispersal. A decrease of genetic diversity towards the southeast suggests that this area has been relatively recently colonized as suggested by isozyme data, pollen records, and ecological niche modelling (**Fig. 7.8**).

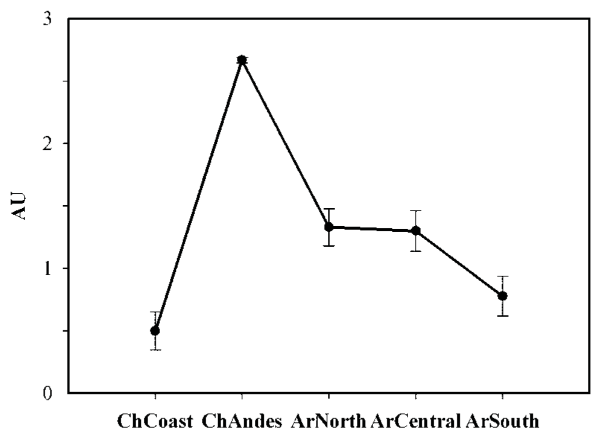

Figure 7.6 Average number of private alleles (AU, i.e. those present in just one population) in five regions, two from Chile (coast and Andes) and three from Argentina (north, centre, and south), along the range of *Austrocedrus chilensis.* Bars represent standard deviations.

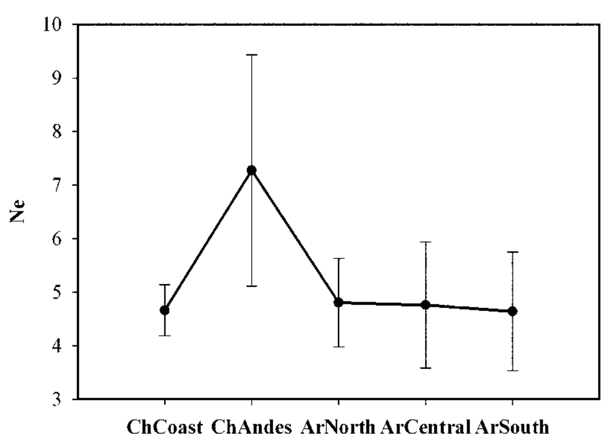

Figure 7.7 Average effective number of alleles (Ne) in five regions, two from Chile (coast and Andes) and three from Argentina (north, centre, and south), along the range of *Austrocedrus chilensis*. Bars represent standard deviations.

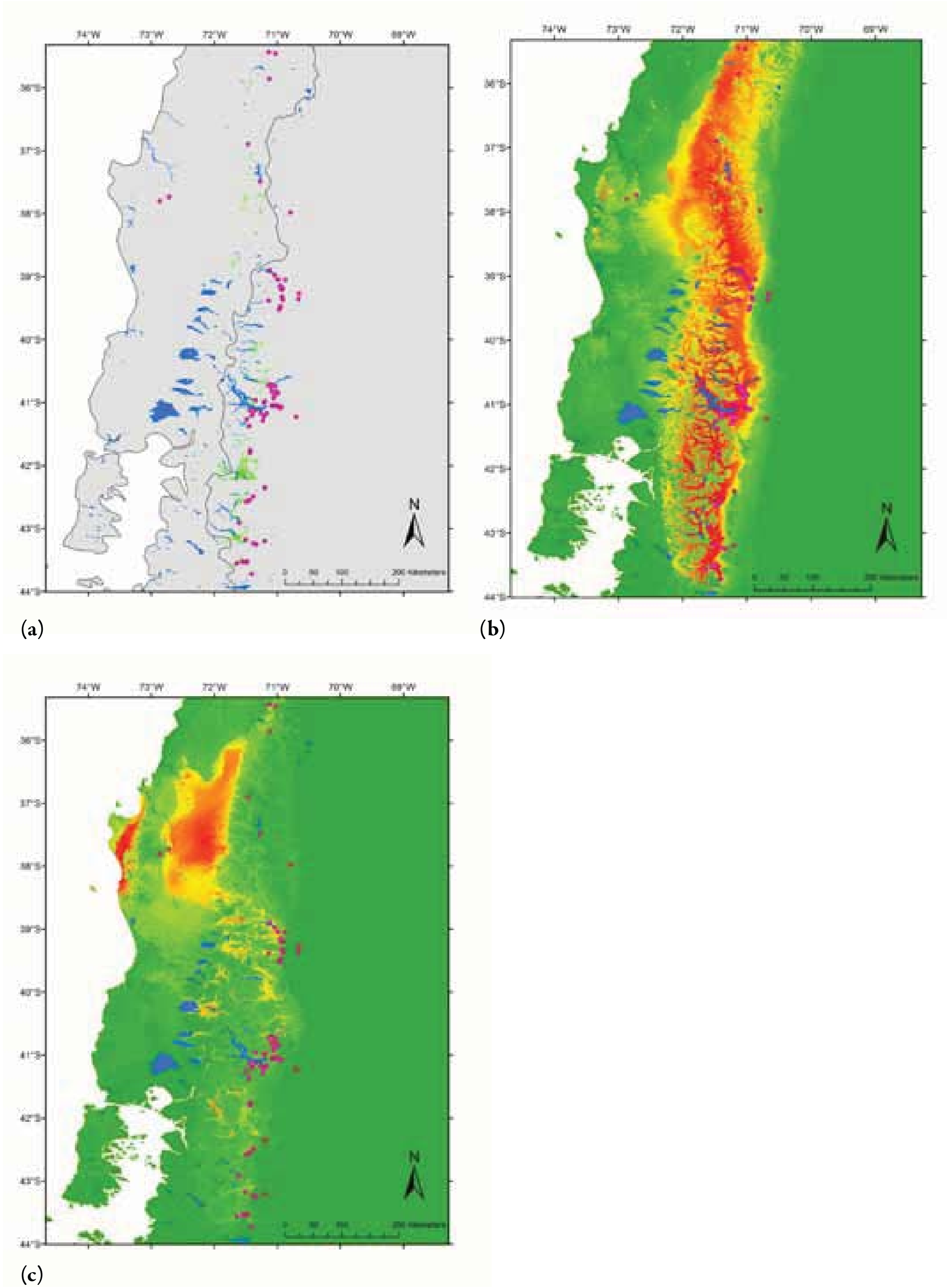

Figure 7.8 Panel (a) depicts the present distribution of *Austrocedrus* in green and sampled populations for genetic studies in pink. Panels (b) and (c) show suitability maps (potential present) and Last Glacial Maximum distribution (past) using ecological niche modelling, respectively.

Conclusions

Forest fragmentation and degradation affect patterns of genetic variation at different scales. The strength of such impacts depends particularly on autoecological characteristics of the studied species, historical events that have shaped gene pools, landscape-use change, environmental gradients, and disturbance regimes that are key determinants of such shifts. Dry forests are currently the focus of increasing conservation and restoration efforts. Restoration recommendations should consider patterns of within- and between-population genetic variation which in turn will be key determinants of restoration success. Specific conclusions for each of the study areas are given below.

(i) Seasonally dry subtropical forests of northern Argentina

Recommendations can be drawn based on previous studies and genetic results on *Cedrela.*

1. Mature stands of *Cedrela balansae* are in urgent need of conservation actions, if they still exist. Additionally, efforts should be made to encourage passive restoration of forest stands in piedmont forests, which in turn are the most threatened due to the advance of agriculture in lowland areas.
2. Active restoration of degraded areas should use germplasm of nearby local populations of *Cedrela* to avoid outbreeding depression, particularly in species with elevated inbreeding.
3. Additional studies are needed to describe the distribution range of *C. saltensis* and its conservation status, i.e. population assessment inside and outside protected areas.
4. Geographic variation patterns of economically relevant quantitative traits and their heritability in *Cedrela* should be investigated. These could be the base for genetic improvement and restoration programmes aiming at developing a resource for sustainable utilization.
5. The combination of distribution patterns at neutral and adaptive traits in the three *Cedrela* species can be used to inform the designation of conservation areas.

(ii) Arid central Oaxaca, México

The genetic effects of fragmentation and forest degradation can be complex and vary considerably among species. Some species can benefit under moderate levels of fragmentation (Ramírez-Luis and del Castillo, 2009; del Castillo *et al.*, unpublished data). Medium and large sized fragments can sustain some species, such as *Catopsis berteroniana*, which only grows at fragment edges. In such conditions, this species can maintain a relatively high genetic diversity without evidence of recent bottlenecks, and demographically shows a nearly stable size distribution and a positive growth (del Castillo, unpublished). There is an urgent need to conduct population studies on fragmented landscapes of temperate dry forests and shrublands of central and southern Mexico.

Some populations of the same species may hold high genetic diversity, as is the case of *Malacomeles denticulata* (Ramírez-Luis and del Castillo, 2009). This implies that breeding programmes should first examine different populations and not generalize *a priori* the genetic characteristics of the species based on studies of one or few populations. Therefore, conservation and restoration programmes should focus at the population level.

(iii) Dry temperate forests of southern Argentina

In the long term, the presence of genetic variation will be the main factor determining the success of restoration actions (Rice and Emery, 2003). Unfortunately it is uncommon to have genetic data available in advance of reintroduction efforts. Patterns of within- population genetic diversity and divergence among different populations across the entire species' range should guide restoration actions. Some recommendations can be drawn from genetic studies on *Austrocedrus*:

Genetic results from nuclear markers show that more genetic diversity exists towards the northern range of *Austrocedrus*. In particular, hypervariable micro-satellite markers provide evidence that northwestern populations in Chile were probably the source of eastern populations located in Argentina. This result is interpreted as postglacial colonization from populations that survived the Last Glacial Maximum on western slopes of the Andes, according to pollen records. Also, ecological niche modelling (**Fig. 7.8**) based on 18 bioclimatic variables and elevation models are also consistent with this hypothesis of range contraction towards northern latitudes for such mesothermic species during LGM conditions. As a result, *Austrocedrus* populations in Chile are of high conservation concern. Urgent conservation actions should be developed to protect remnant stands of *Austrocedrus* in those highly disturbed and genetically diverse areas and also to promote passive restoration.

Northern naturally fragmented populations in Argentina have elevated genetic diversity a result of early postglacial colonization and therefore are of great conservation value. Efforts should be devoted to facilitate expansion of those populations. We recommend the exclusion of exotic cattle to promote passive restoration within those fragments such that natural establishment occurs.

Genetic diversity of *Austrocedrus* is geographically structured. Recommendations based on suitability maps for *Austrocedrus* have identified eight potential areas for restoration (**Fig. 7.9**). Given that *Austrocedrus* populations show latitudinal differences in genetic traits, and that potential areas to be restored are concordant with genetic structure, the design of restoration practices should include local germplasm collection and propagation.

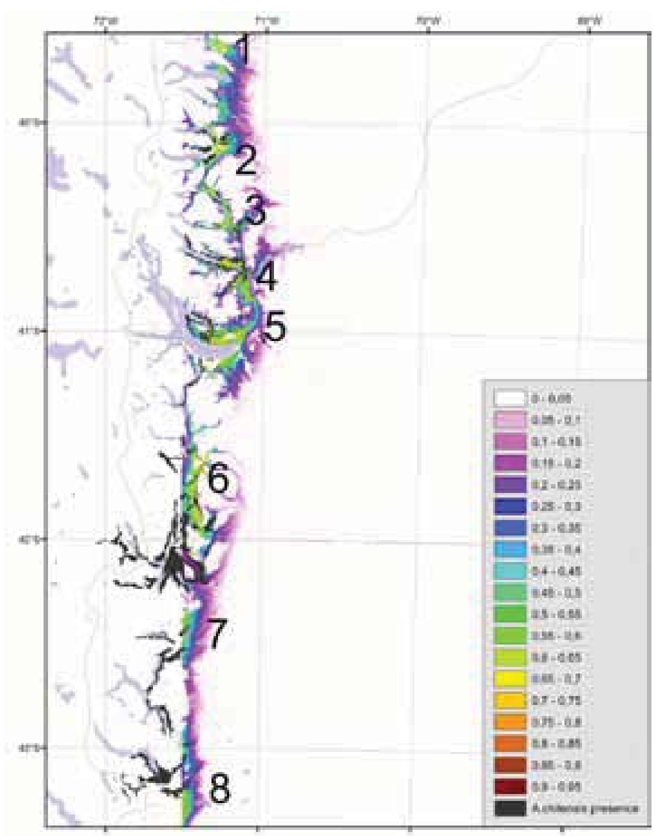

Figure 7.9 Suitability of *Austrocedrus* (colour scale) and identification of areas to be restored (numbers). The suitability map was developed by training a bioclimatic model based on 18 variables and elevation using the current distribution of the species (dark gray).

In conclusion, genetic data on *Austrocedrus* highlight the importance of historical and current processes when implementing conservation actions and restoration procedures. Relevant guidelines for restoration need to be species-specific and should also consider genetic patterns such that local germplasm is used in restoration efforts in species with significant genetic structures.

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8 LANDSCAPE-SCALE DYNAMICS AND RESTORATION OF DRYLAND FOREST ECOSYSTEMS

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Introduction

The restoration of forest landscapes is typically achieved either through passive restoration, involving the establishment of forest cover through natural regeneration, or some form of active restoration approach, involving the establishment of trees by artificial means. The existence of different restoration options raises the question of how an appropriate restoration approach might best be identified for any individual location. Where feasible, passive restoration approaches are often likely to be preferred, because of the intrinsically lower costs of tree establishment. However, the potential for natural regeneration of forest cover may often be limited, particularly in landscapes that are highly degraded. Factors that can limit the process of natural regeneration include a lack of a source of propagules, perhaps because an individual site is isolated from remnant forest stands; adverse site characteristics for seed germination or seedling establishment, such as degraded or compacted soils (see Chapter 5); or an adverse disturbance regime that causes high mortality of juvenile trees. In areas where passive restoration approaches are associated with a high risk of failure, active restoration approaches may be preferred.

Restoration planning should therefore ideally be informed by an understanding of where natural regeneration is likely to occur on a landscape, within a given timescale. Such information might help identify those locations where natural regeneration is most likely to occur under different patterns of land use. In order to make such predictions, some form of forest modelling approach is required. Ideally, such a model should incorporate those ecological processes that influence natural regeneration, including dispersal, competition, and tree survival and growth in locations with different site characteristics. In addition, it is essential that the model enables predictions or forecasts to be made that are spatially explicit, in order to be of practical value in restoration planning.

As noted by Newton (2007a), a wide variety of different forest modelling approaches have been developed, although only a small proportion of these are suitable for exploring forest dynamics at the landscape scale. Spatially explicit approaches to forest modelling have been greatly supported by recent developments in Geographic Information System (GIS) technologies, which now enable GIS to be linked to models of forest dynamics, either for pre-processing data for use in a non-spatial modelling, or for displaying model output. Closer linkages between the model and GIS can be achieved if they share the same data structures. Examples of models that have been developed specifically to operate at landscape scales are provided by Frelich *et al.* (1998), Frelich and Lorimer (1991) and Liu and Ashton (1998).

However, the use of forest models to support restoration planning, or conservation management more generally, has been very limited to date (Newton, 2007a). Such models have enormous potential for supporting decision making relating to sustainable forest management, particularly if they can be linked with other analytical tools, such as GIS (Newton *et al*., 2009a).

In this chapter, we describe the application of a spatially explicit model to examine the dynamics of dryland forest landscapes, with the aim of informing forest restoration planning. Specifically, the objective of this investigation was to simulate the landscape-scale dynamics of dryland forest in order to assess the potential for natural recovery of forest landscapes under different disturbance regimes. Information on the rate of ecological recovery under different disturbance scenarios is required to evaluate the feasibility of passive restoration approaches (Vieira and Scariot, 2006). Ideally such information should be spatially explicit, given that forest restoration should be undertaken at the landscape scale in order to address the problem of forest fragmentation and to restore connectivity (Mansourian *et al.,* 2005). Although a number of studies have analyzed dry forest recovery after disturbance (Guariguata and Ostertag, 2001; Vieira and Scariot, 2006; Sampaio *et al.,* 2007; Griscom *et al.,* 2009), very little information is available regarding the processes of recovery of dry forest at the landscape scale.

This study employed the model LANDIS-II, which was designed to simulate the dynamics of forested landscapes through the incorporation of ecological processes including succession, disturbances and seed dispersal over long time domains (Scheller *et al.,* 2007). The LANDIS-II model is an elaboration of the LANDIS family of landscape disturbance and forest succession models. Although the architecture has changed since the initial version (Mladenoff *et al.,* 1996) and new features have been added, LANDIS-II retains many principles from earlier versions that have been widely tested and applied in different parts of the world (He and Mladenoff, 1999; Mladenoff and He, 1999; Mladenoff, 2004; Scheller *et al.,* 2005; Wang *et al.,* 2006; Swanson, 2009). However, we are not aware of any previous attempt to apply LANDIS-II, or any other spatially explicit model of forest dynamics, to dry forest landscapes in Latin America.

This chapter first provides a brief description of the LANDIS-II model, and then presents an overview of the results obtained from applying LANDIS-II to four different study areas in Latin America; two in Mexico, one in central Chile and one in southern Argentina. The implications of the results obtained for planning the restoration of dryland forest landscapes are then explored.

The LANDIS-II model

The LANDIS model is described in more detail elsewhere (He and Mladenoff, 1999; Mladenoff and He, 1999; Mladenoff, 2004); <http://www.landis-ii.org>). In essence, LANDIS-II uses a cell-based data format; within each cell it tracks the presence/absence of tree species age cohorts at a time step specified by the user. Vegetation patches can aggregate and disaggregate in response to spatial patterns of stochastic rules of disturbance and succession. Tree species succession is a competitive process governed by species life history parameters (longevity, age of sexual maturity, shade and fire tolerance class, effective and maximum seed dispersal distance, vegetative reproduction probability, minimum and maximum age of vegetative reproduction, and post-fire regeneration), and the probability of species establishment on different ecoregions (or landtypes). Tree succession interacts with several spatial processes (i.e. seeding, wind and fire disturbances, and harvesting). As disturbance in LANDIS-II is stochastic, calibration is required to ensure the output is fitted to the ecological values of the simulated area (Franklin *et al.,* 200;, Lafon *et al.,* 2007; Syphard *et al.,* 2007).

Case studies in Mexico

Study areas

Research was undertaken in two study areas dominated by tropical dry forest (TDF), namely the Tablon, Chiapas, and central Veracruz, Veracruz, Mexico. Both study areas are global conservation priorities, having been identified as global biodiversity hotspots (Myers *et al.,* 2000), and in recent decades both have been degraded at a high rate owing to the effects of human disturbances (Challenger and Dirzo, 2009). Both study areas cover similar areas, but they differ in the percentage of forest cover. While in central Veracruz only 27% of the land-cover of the entire area is represented by forests and shrubland, almost 90% of the land-cover of Tablon is represented by a forest type. According to CONABIO (2006) both areas have a high degree of marginalization; an average of 23 inhabitants km^2 and 14 inhabitants km^2 respectively were recorded in 2000 in Central Veracruz and Tablon.

The Tablon study area covers 24,735 ha and is situated between 675 and 1537 m altitude in the municipalities of Villaflores and Jiquipilas, state of Chiapas (16º11'38" and 16º22'29" N, and 93º31'57" and 93º44'31" W). The climate is defined as warm sub-humid, with an average annual rainfall between 1200 and 2800 mm concentrated from late May to early November (Aguilar-Jiménez, 2008). The natural vegetation of Tablon forms a gradient of forest types ranging from low-stature deciduous tropical forest in the lower elevations of the study area, through dry oak and pine-oak with increasing elevation, and with pine forests on the highest ridges. Tablon falls within the La Sepultura Biosphere Reserve, which was designated in 1995 for its high number of endemic species, high biodiversity value and species richness (**Box 8.1**).

Box 8.1 Anthropogenic impact on dry forests in Chiapas, Mexico

N. Tejedor

Tropical dry forests, which host a large number of endemic species (Mooney *et al*., 1995), are among the least studied forests worldwide and the most threatened by human actions (Miles *et al*., 2006). La Sepultura Biosphere Reserve, which is located in the southwestern region of the state of Chiapas, Mexico, between 16°00'18" and 16°29'01" of north latitude and 93°24'34" and 94°07'35" west longitude, with an area of 167,309 ha, has been designated for its large amount of endemic species and its various examples of ecosystems scarcely represented in other protected areas of Mexico, including areas of dry forests (INE,1999).

Forest fires are an important problem in Chiapas, and in the reserve this situation is worsened by the fact that the region is extremely vulnerable during the dry season (November to April, and even May) owing to the strong winds from the coast (October to March), which dry the grass and increase the likelihood of fast propagation of fires (Hernandez-Lopez, 2005). The impact of fires on the existing vegetation is exacerbated by agricultural practices such as maize cultivation using the slash and burn technique, which increases soil fertility for a period of 3 or 4 years until the soil nutrients are exhausted, leading to further forest fires and deforestation. This is despite the fact

Slash and burn agriculture in a tropical subdeciduous forest in Ocuilapa, Chiapas, México. Photo: N. Ramirez-Marcial

Reforestation of seasonal dry premontane forest with timber species. Photo: L. Malizia

Forest restoration in progress in Chile. Photo: C. Echeverria

Box 8.1 (cont.)

that there are restrictions on the use of fire as a mechanism to convert the forest to pasture land or control of forest succession, as mentioned in the management plan (INE, 1999) **(Figs. 1** and **2**). According to the geographical database in LAIGE (2007), land-use cover changes have been identified between 1975 and 2000, alongside roads and around population centres, where forests have been converted to agriculture and pasture land.

Areas within La Sepultura, as with many other areas within Chiapas (e.g. Hammond, 1995), have highly compacted soils and the sources of propagules are depleted as a result of anthropogenic disturbance, allowing only the colonization of ruderal species. For example, in a survey carried out in 2007 (Tejedor, 2007) at 1065 m a.s.l within the reserve, where species such as *Quercus sp*., *Phoebe chiapensis*, *Inga sp*. and *Manilkara zapota,* are expected to be found, the only species present from those mentioned were *Quercus sapotifolia* and a species that grows favourably in disturbed areas and savannah type vegetation, *Byrsonima crassifolia* (**Fig. 3**). The dominant vegetation was the ground vegetation, which was mainly composed of grass species. The canopy cover was 43.84%. This suggests that the main disturbance in the area is the conversion of forest for grazing under mature trees, through fire and cutting. A natural disturbance is soil erosion, as the area is on a slope; but this is exacerbated by the deforestation which leads to lack of soil stability.

Figure 1 Fire in La Sepultura biosphere reserve. Photo: N. Tejedeor

Box 8.1 (cont.)

Figure 2 Fire in La Sepultura biosphere reserve. Photo: N. Tejedeor

Figure 3 Surveyed area in La Sepultura biosphere reserve. Photo: N. Tejedeor

The central Veracruz study region, with an area of 29,468 ha, is situated between 10 and 507 m altitude in the state of Veracruz, Mexico (19º07'45" and 19º21'18"N, and 96º21'33" and 96º41'12"W). The climate is defined as warm sub-humid (minimum and maximum average temperatures are 20° and 31°C, respectively) with rainfall of 800 to 1500 mm, occurring primarily from June to September followed by an extended dry season. Areas on the eastern side of central Veracruz have a warm-humid climate, whereas those on the western side are characterized by a warm-dry climate. The original vegetation was predominantly tropical dry forest. None of the remaining forest fragments are under protection. The primary land use is cattle ranching, which is generally undertaken on a relatively small scale by private landowners.

Model parameterization

The LANDIS-II model is designed to accept raster imagery as a spatially explicit input to simulate landscape dynamics. Full details of model parameterization for these study areas are presented by Cantarello *et al.* (2010). In Tablon, input raster data included a Digital Elevation Model (DEM) and QuickBird satellite imagery, from which a series of secondary maps were derived. The DEM (50 m cell-size) was derived from the 30 m resolution national DEM, resampled to a 50 m grid using regularized spline with tension (Mitasova and Mitas, 1993). A direct beam solar radiation map (50 m cell-size) was calculated using the formulae proposed by Rigollier *et al.* (2000) and implemented in the GRASS module r.sun (Neteler and Mitasova, 2008). An ecoregions map was produced from the combination of the DEM with the beam solar radiation map.

Three QuickBird scenes acquired in November–December 2004 were obtained as a mosaic to cover the study area. A basic land-cover map (50 m cell-size), which identified forest, pasture, roads, urban areas and permanent agricultural areas was derived from the QuickBird imagery. The production of the basic land-cover map (**Fig. 8.1**) involved a combination of supervised classification of the multispectral mosaic to separate forest from pasture with the manual digitizing of roads, urban areas and permanent agricultural areas on the pan mosaic. A forest stand types map (50 m cell-size) was derived from the QuickBird imagery and ecoregions map. The production of the forest stand types map involved performing an unsupervised classification of the forest component of the QuickBird imagery into twenty forest classes which were split by ecoregions into a 28-forest stand types layer. This layer was validated by manual comparison against the panchromatic mosaic and the field survey data describing species and stem diameters distributions.

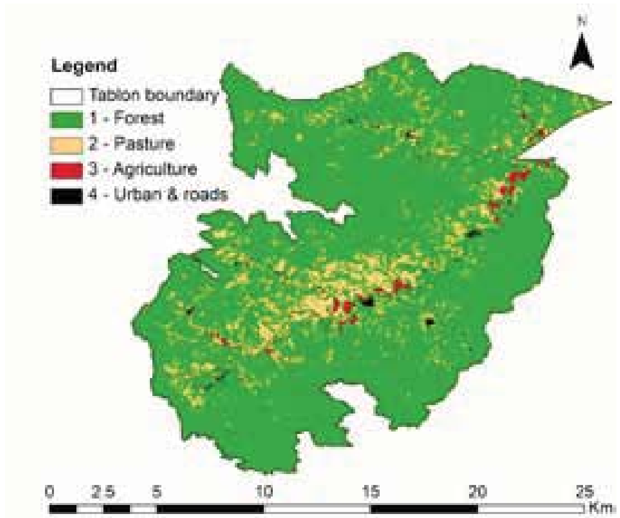


Figure 8.1 Land-cover map for the Tablon study area produced from QuickBird imagery classification. The forest cover includes 28 forest types, not displayed here.

In central Veracruz, input raster data included a DEM and SPOT satellite imagery, which were used to obtain a series of secondary maps. The DEM (cell-size 80 m) was derived from Shuttle Radar Topography Mission (SRTM) data with a resolution of three arc seconds per pixel. A soil types map (cell-size 80 m) was extracted from the national edaphology map (INIFAP and CONABIO, 1995). An ecoregions map was produced from the combination of the DEM with soil types map.

Three SPOT high-resolution visible and infrared (HRVIR) multispectral images (20 m resolution) from December 2007 and January 2008 were used to produce a land-cover map (80 m cell-size), which identified shrubland, forest, agriculture, urban and roads, water, plantation and pasture. Validation was conducted through field visits and visual inspection of high resolution Google Earth images (©2008 Google Inc., California, USA). A forest stand types map (80 m cell-size) was derived from the land-cover map and the field survey data (**Fig. 8.2**).

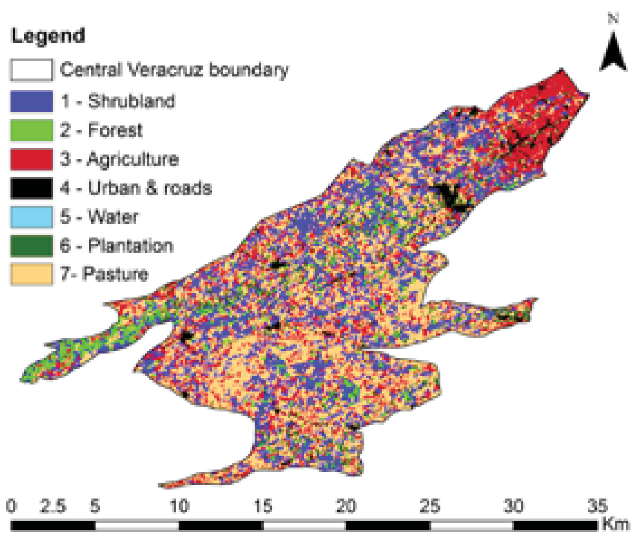
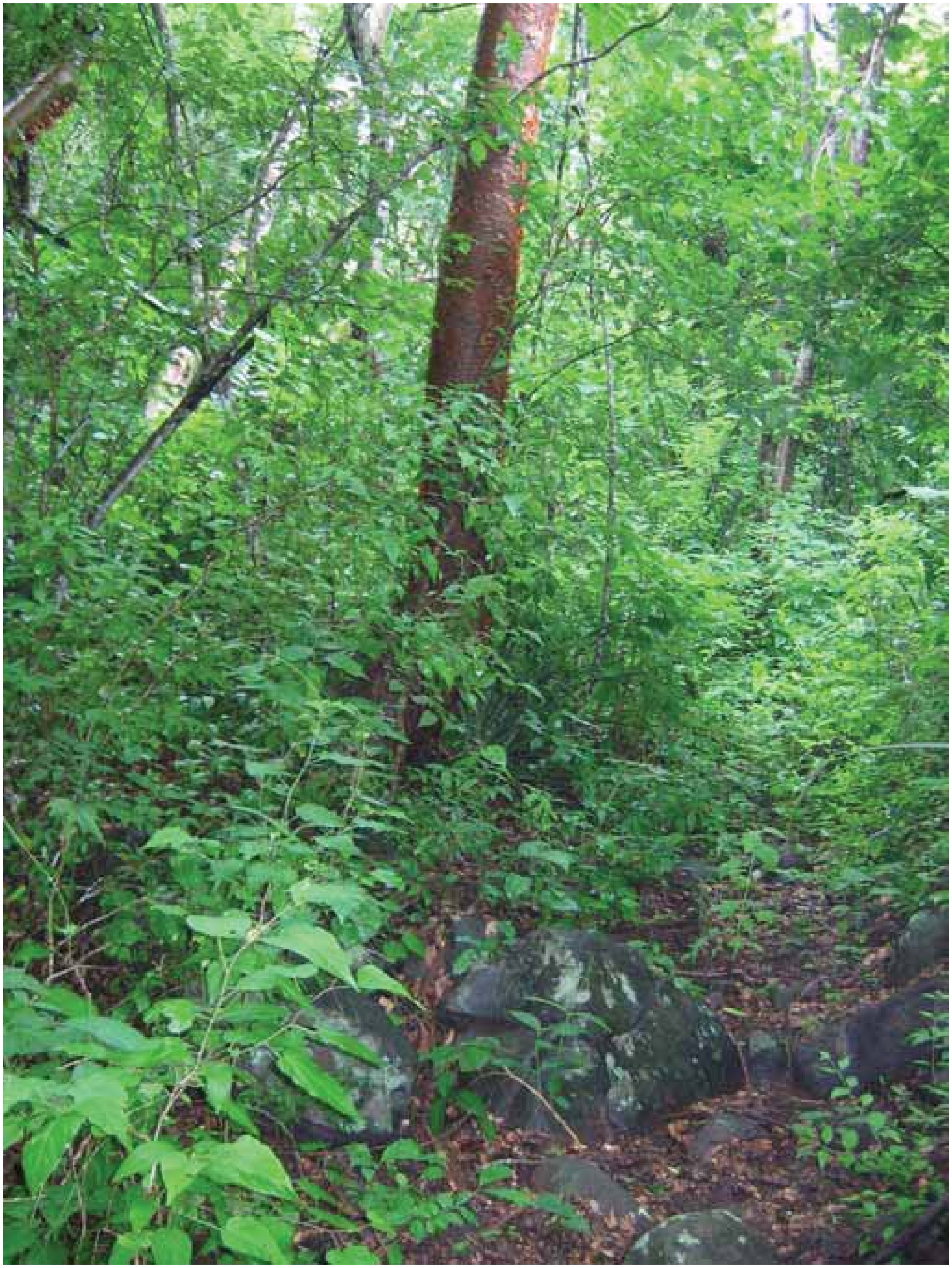


Figure 8.2 Land-cover map for the central Veracruz study area produced from SPOT 4 imagery classification). The forest cover includes 15 forest types, not displayed here.



***Bursera simaruba*, one of the dominant tree species in the tropical dry forest of Paso de Ovejas, Veracruz. Photo taken in Hato Los Marines during the rainy season. Photo: G. Williams-Linera**

To provide field data, 36 circular plots (0.1 ha each) were established along an altitudinal gradient in Tablon and 100 survey plots (0.01 ha each) were established within ten TDF fragments in central Veracruz (Williams-Linera and Lorea, 2009). In each plot, tree species were identified and diameter at breast height (dbh) of each tree >5 cm was measured. Dbh values were converted into ages by consulting a group of local experts.

In LANDIS-II the landscape is stratified into ecoregions, which are ecologically homogenous sub-areas characterized by the same habitat suitability (establishment probability) for each species to be modelled. Ecoregions can be active or non-active depending on whether they represent areas where forests can grow or not. In Tablon, one non-active and fifteen active ecoregions were considered; active ecoregions covered 24,354 ha, non-active 381 ha. The species establishment probabilities for each ecoregion were derived from the outputs of generalized additive models as interpreted by local experts. Input to the models included species occurrence data, obtained from the MOBOT Tropicos® data base (©2010 Missouri Botanical Garden, USA), and climate data extracted from the Worldclim database (Hijmans *et al.,* 2005). In central Veracruz, one non-active and thirteen active ecoregions were considered; active ecoregions covered 20,788 ha, and non-active 8,680 ha. The species establishment probabilities for each ecoregion were derived from the outputs of GARP models (Stockwell and Peters, 1999), which were produced by using the software DesktopGarp (©2002 University of Kansas Center for Research, Inc., USA) and interpreted by local experts.

The inputs required by LANDIS-II include information about the distribution, composition and age structure of forest stands at year 0. In both study areas, to populate each cell of the landscape with species and age cohort we combined the forest cover map with the field survey data describing species and age distributions.

Species attributes

For each species to be modelled, LANDIS-II requires information about longevity, age of sexual maturity, shade and fire tolerance class, effective and maximum seed dispersal distance, vegetative reproduction probability, minimum and maximum age of vegetative reproduction, and post-fire regeneration. In central Veracruz, species attributes were extracted for the 22 most abundant species (i.e. species with >14 sampled individuals in the field survey). The subset was characterized by a range of shade and fire tolerances (**Table 8.1**). In Tablon, none of the 23 most abundant species (i.e. species with >5 sampled individuals in the field survey) had high shade tolerance. Two relatively infrequent species (*Sapindus saponaria* and *Gyrocarpus mocinnoi*) were therefore included in the model, to ensure that the full range of shade tolerance was included, giving a total of 25 species included in the model. In both study areas species attributes were extracted from the scientific literature and by consulting local experts.

Table 8.1 Details of the species characteristics in central Veracruz. Long: longevity (years); Mat: age of maturity (years); ShT: shade tolerance class (1 to 5, with 1 for the most shade intolerant and 5 for the most shade tolerant); FiT: fire tolerance class (1 to 5, with 1 for the least tolerant and 5 for the most tolerant); EffSD: effective seeding distance (m); MaxSD: maximum seeding distance (m); VRP: vegetative reproduction probability; MinVRP: minimum age of vegetative reproduction (years); MaxVRP: maximum age of vegetative reproduction (years); P-FiR: post-fire regeneration (form of reproduction that the species adopts after fire events).

Species	Species abbr.	Long	Mat	ShT	FiT	EffD	MaxSD	VRP	Min VRP	Max VRP	P-FiR
Acacia cochliacantha	Acaccoch	50	2	1	5	5	100	1	2	50	re-sprout
Brosimum alicastrum	Brosalic	150	20	5	1	5	10000	1	3	60	re-sprout
Bursera cinerea	Burscine	80	5	4	1	5	10000	0	0	0	none
Bursera fagaroides	Bursfaga	80	5	4	3	5	10000	0	0	0	none
Bursera graveolens	Bursgrav	80	5	4	2	5	10000	0	0	0	none
Bursera simaruba	Burssima	80	5	3	3	5	10000	1	2	50	re-sprout
Calyptranthes schiedeana	Calyschi	60	15	5	2	5	10000	0	0	0	none
Cochlospermum vitifolium	Cochviti	60	15	3	5	3	100	1	1	30	re-sprout
Ceiba aesculifolia	Ceibaesc	40	10	3	2	5	100	1	2	30	re-sprout
Comocladia engleriana	Comoengl	70	25	3	2	200	1000	1	7	60	re-sprout
Eugenia hypargyrea	Eugehypa	40	5	5	1	5	100	0	0	0	none
Guazuma ulmifolia	Guazulmi	40	3	1	5	5	100	1	2	40	re-sprout
Heliocarpus donnell-smithii	Helidonn	50	10	1	3	100	100	1	3	25	re-sprout
Ipomoea wolkottiana	Ipomwolc	60	10	1	3	11	100	1	2	20	re-sprout
Leucaena lanceolata	Leuclanc	20	2	1	2	5	100	0	0	0	serotiny
Luehea candida	Luehcand	70	20	3	2	20	100	1	3	60	re-sprout
Lysiloma microphyllum	Lysimicr	70	15	1	2	5	100	1	3	60	re-sprout
Savia sessiliflora	Savisess	30	5	5	1	5	100	0	0	0	none
Senna atomaria	Sennatom	30	5	1	4	5	100	1	2	10	re-sprout
Stemmadenia pubescens	Stempube	30	5	3	1	5	1000	0	0	0	none
Tabebuia chrysantha	Tabechry	120	15	3	3	20	100	1	2	50	re-sprout
Thouinidium decandrum	Thoudeca	70	10	1	4	20	100	1	3	60	re-sprout

Table 8.2 Details of the species characteristics in Tablon. Long: longevity (years); Mat: age of maturity (years); ShT: shade tolerance class (1 to 5, with 1 for the most shade intolerant and 5 for the most shade tolerant); FiT: fire tolerance class (1 to 5, with 1 for the least tolerant and 5 for the most tolerant); EffD: effective seeding distance (m); MaxSD: maximum seeding distance (m); VRP: vegetative reproduction probability; MinVRP: minimum age of vegetative reproduction (years); MaxVRP: maximum age of vegetative reproduction (years); P-FiR: post-fire regeneration (form of reproduction that the species adopts after fire events).

Species	Species abbr.	Long	Mat	ShT	FiT	EffD	MaxSD	VRP	Min VRP	Max VRP	P-FiR
Acacia cornigera	Acaccorn	30	3	1	4	50	10000	1	1	15	re-sprout
Acacia pennatula	Acacpenn	40	3	1	4	100	10000	1	1	20	re-sprout
Bursera bipinnata	Bursbipi	50	5	3	3	25	10000	1	3	25	re-sprout
Bursera excelsa	Bursexce	50	5	3	3	20	10000	1	3	25	re-sprout
Bursera simaruba	Burssima	80	3	3	4	20	10000	1	3	30	re-sprout
Byrsonimia crassifolia	Byrscras	30	5	1	5	10	15000	1	3	15	re-sprout
Diphysa robiniodes	Diphrobi	40	8	2	3	20	100	1	3	40	re-sprout
Erythrina chiapasana	Erytchia	40	5	2	3	10	100	1	3	20	re-sprout
Erythrina folkersii	Erytfolk	40	6	1	3	10	100	1	3	25	re-sprout
Eysenhardtia adenostylis	Eyseaden	30	4	2	1	20	100	1	1	30	re-sprout
Guazuma ulmifolia	Guazulmi	30	3	1	5	100	1000	1	2	25	re-sprout
Gyrocarpus mocinnoi	Gyromoci	30	4	5	4	10	100	0	0	0	none
Heliocarpus reticulatus	Helireti	40	10	1	1	20	100	0	0	0	none
Leucaena diversifolia	Leucdive	25	3	1	2	10	10	1	1	20	re-sprout
Lonchocarpus rugosus	Loncrugo	45	10	2	3	20	15000	1	5	25	re-sprout
Pinus maximinoi	Pinumaxi	100	20	1	4	20	100	0	0	0	none
Pinus oocarpa	Pinuooca	100	7	1	5	20	100	0	0	0	serotiny
Quercus acutifolia	Queracut	120	20	2	4	20	10000	1	5	60	re-sprout
Quercus castanea	Quercast	80	15	2	2	10	10000	1	5	40	re-sprout
Quercus conspersa	Quercons	120	20	2	4	20	10000	1	5	60	re-sprout
Quercus elliptica	Querelli	200	20	2	5	100	10000	1	1	80	re-sprout
Quercus penducularis	Querpedu	200	15	2	5	100	10000	1	1	80	re-sprout
Quercus segoviensis	Quersego	200	20	2	4	100	10000	1	1	80	re-sprout
Sapindus saponaria	Sapisapo	80	15	4	1	10	10	0	0	0	none
Ternstroemia tepezapote	Terntepe	40	5	1	3	100	10000	1	1	30	re-sprout

Table 8.3 Mean simulated forest area after 400 years in two Mexican study areas, under different disturbance scenarios. For details, see text.

	Tablon		Central Veracruz	
	$\bar{x}$	S.D.	$\bar{x}$	S.D.
NO-DIST	91.24	6.59	99.89	0.14
GRAZ	80.69	10.2	99.14	0.36
SIF	88.67	3.72	92.91	0.40
SIF-GRAZ	81.16	5.31	92.36	0.48
LFF	87.00	3.13	59.74	4.19
LFF-GRAZ	76.28	6.04	21.79	4.95

Disturbance scenarios

Scenarios were developed to explore the impacts of both fire and grazing, as these disturbances have caused the most serious degradation of dry forest in Mexico. Six different scenarios were simulated: (i) no disturbance (NO-DIST); (ii) grazing without fire (GRAZ); (iii) small, low intensity, infrequent fires without grazing (SIF); (iv) small, low intensity, infrequent fires with grazing (SIF-GRAZ); (iv) large, intense, frequent fires without grazing (LFF); and (vi) large, intense, frequent fires with grazing (LFF-GRAZ). The Base Fire (v2.1) and Base Harvest (v1.2) extensions of LANDIS-II were used to generate the fire and grazing scenarios. LANDIS simulations were conducted for 400 years to allow fire and grazing to fully manifest their effects on forest succession. The time steps were set at 10 years for tree succession, 5 years for fire disturbance and 1 year for grazing.

Results

The LANDIS-II outputs consist of raster maps, each corresponding to a time step specified by the user (10 years in this study). Forest cover (defined here as percentage cover of trees >10 years) increased quite rapidly under NO-DIST scenario from 89.9% to 97.6% in Tablon and 6.26% to 99.6% in central Veracruz after only 50 years, producing a landscape dominated by forest cover after 400 years (**Figs. 8.3** and **8.4**). Under LFF, SIF, GRAZ and SIF-GRAZ scenarios, forest cover still increased, but values were lower than the NO-DIST scenario after 50 years, with values reaching between 90.1% and 96.2% in Tablon; and 50.8% and 99.5% in central Veracruz. Under LFF-GRAZ scenario the forest cover decreased after 50 years to 86.3% in Tablon, but slightly increased to 9.62% in central Veracruz. Differences in mean forest cover were recorded under the six scenarios in both study areas, with lowest values recorded under LFF-GRAZ in both cases.

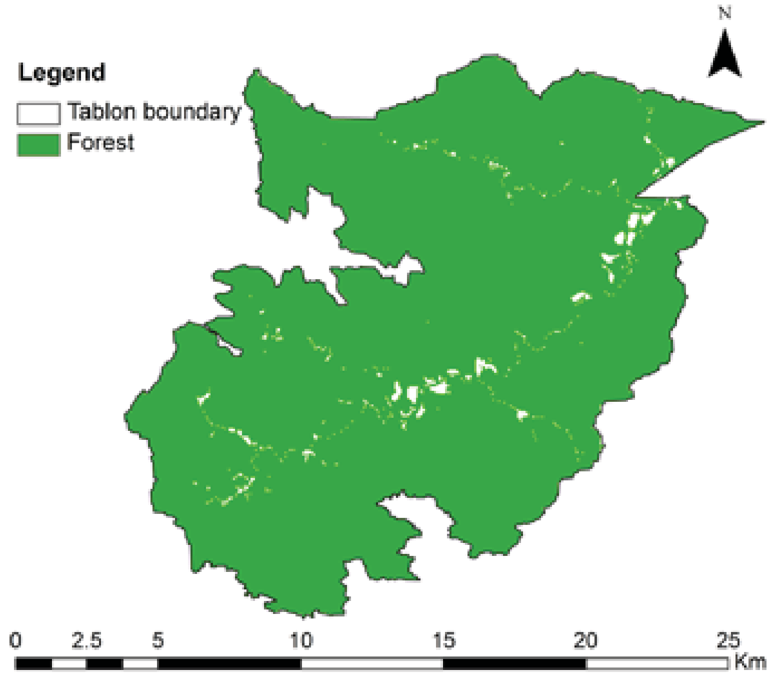


Figure 8.3 Map to illustrate projected forest cover after 400 years in Tablon.

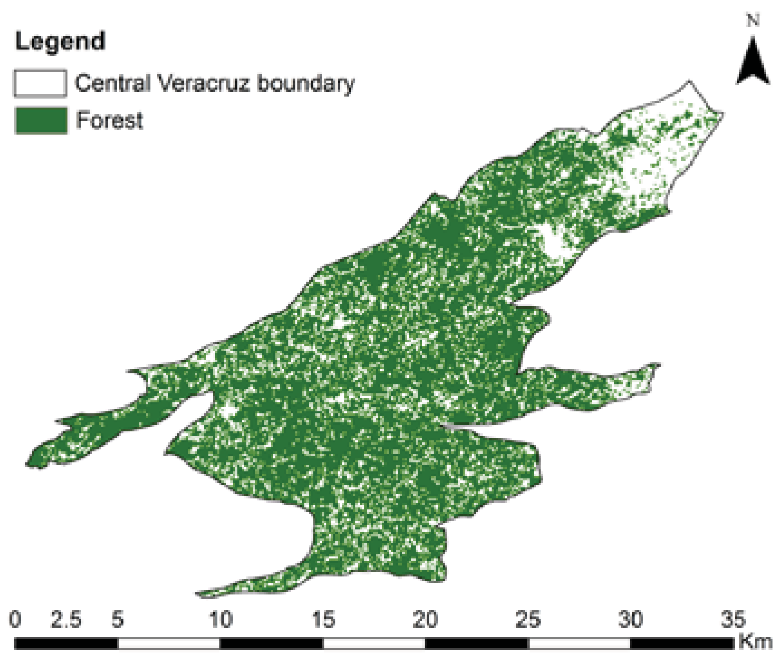


Figure 8.4 Map to illustrate projected forest cover after 400 years in central Veracruz.

Simulation results indicated that tree size class structure differed under the six scenarios in both study areas. Over 400 years, the highest percentage cover of the landscape was occupied by: (i) trees >100 years old under NO-DIST, SIF, GRAZ and SIF-GRAZ scenario after 100 years (73.9%, 46.5%, 72.3%, and 41.1%, respectively); and (ii) trees between 31–60 years old under LFF and LFF-GRAZ scenarios after 50 years (45.0% and 35.1%, respectively). When only the fire and grazing disturbance scenarios are considered, the median percentage cover of trees older than 60 years was the highest under the SIF scenario (53.0%), which differed statistically from the median percentage cover value under the LFF-GRAZ scenario (p<0.05; Mann-Whitney test). In central Veracruz, similar results to Tablon were obtained when median percentage cover of trees > 60 years was compared. Values were the lowest under the LFF-GRAZ scenario (0.05%), which differed statistically from the median percentage cover values under all of the fire and grazing scenarios (p <0.05 in each case; Mann-Whitney test). When only the fire and grazing disturbance scenarios are considered, the median percentage cover of trees older than 60 years was the highest under the SIF scenario (64.6%), which differed statistically from the median percentage cover values under the LFF, SIF, LFF-GRAZ and SIF-GRAZ scenarios (p <0.05 in each case; Mann-Whitney test).

Simulation results show that modelled species richness (25 modelled species out of 53 sampled species in Tablon, and 22 modelled species out of 97 recorded species in central Veracruz) differed under the six scenarios in both study areas. In Tablon the median percentage area with ≥5 species was the highest under the SIF scenario (47.8%), which differed statistically from the median percentage cover values under the NO-DIST and GRAZ scenarios (p <0.05 in each case; Mann-Whitney test). When only the fire and grazing disturbance scenarios were considered, the median percentage area with ≥5 species was the lowest under the GRAZ scenario (30.4%), which differed statistically from the median percentage cover values under the LFF, SIF and LFF-GRAZ scenarios (p <0.05 in each case; Mann-Whitney test). In central Veracruz at year 0, similar to Tablon, the median percentage area with ≥5 species was the highest under the SIF scenario (31.5%), which differed statistically from the median percentage cover values under all of the fire and grazing scenarios (p <0.05 in each case; Mann-Whitney test). When only the fire and grazing disturbance scenarios were considered, the median percentage area with ≥5 species was the lowest under the LFF scenario (0.14%), which differed statistically from the median percentage cover values under the SIF, GRAZ and SIF-GRAZ scenarios (p <0.05 in each case; Mann-Whitney test).

Discussion

The LANDIS-II simulations performed in this study suggest that forest cover can increase quite rapidly if protected from grazing and fire, a finding that is supported by field observations. For example, Sampaio *et al.* (2007) indicated that TDF in central Brazil was able to recover rapidly in the absence of anthropogenic disturbance, owing to the re-sprouting ability of the tree species present in that area. Griscom *et al.* (2009) noted that in Panama, relatively diverse second-growth dry forests were produced within three years following pasture abandonment. Results obtained here also suggest that with time, an increasing proportion of the landscape becomes occupied by trees older than 60 years. This is in accordance with the observation made by Powers *et al.* (2009) indicating that in regenerating dry forests in Costa Rica, larger individuals increase asymptotically with stand age. These results highlight the potential scope for passive restoration of TDF, if forests can be adequately protected, and imply that such forests may be relatively resilient to human disturbance.

Simulations indicate opposite trends with regards to species richness in the two study areas considered here, with the number of species tending to increase over time in central Veracruz but to decline in Tablon, in the absence of disturbance. The majority of the species present initially were highly fire-tolerant but relatively intolerant of shade, and are therefore likely to disappear as a result of competitive processes as the canopy closes. Simulations also show that in both study areas in the absence of disturbance, the landscape is increasingly occupied by shade-tolerant and re-sprouting species with a relatively long seed dispersal distance; these species cover between 76–99% of the landscape after 200 years. These results are consistent with other studies that suggest shade-tolerant species tend to be dominant in secondary forests after 100–400 years since abandonment (e.g. Guariguata and Ostertag 2001).

The LANDIS-II simulations under both a scenario of small, surface, low intensity infrequent fires (SIF) and a scenario of large, crown, high-intensity frequent fires (LFF) suggest that forest cover can recover quite quickly in terms of extent, owing to the high re-sprouting ability of the tree species. However under LFF, in Tablon, the forest cover started to decline gradually after 50 years; in central Veracruz, tree species older than 10 years only occupied half of the cover value under the LFF scenario than under SIF. In contrast with the SIF scenario, the LFF scenario explored here strongly modified the forest structure and composition in both Tablon and central Veracruz, highlighting an issue of concern regarding the potential impacts of increasing frequency of high intensity fires on dry forest conservation in this region.

The LANDIS-II projections reveal that when grazing is acting in combination with fire, the forest cover, structure and composition vary markedly depending on the intensity, frequency and extent of the fires. The forest cover under LFF-GRAZ was lower than under LFF alone in both study areas, suggesting that grazing significantly affects the forest cover under a frequent, high intensity fire regime. The combination of frequent, high intensity fires and grazing was also found to negatively affect the structure and composition of dry forest in both study areas. These results are consistent with other studies that indicate repeated fire and grazing as the most important types of anthropogenic disturbance impeding TDF regeneration (Janzen, 1988).

These results suggest that passive restoration of dry forest is achievable at the landscape scale in both Tablon and central Veracruz if grazing were to be excluded, and fire were to be carefully managed to achieve a regime of infrequent, low intensity fires. Of the four disturbance scenarios analyzed, SIF is the only scenario that does not negatively affect forest cover, composition and structure of TDF. Simulations revealed that a combination of frequent, intense fires and grazing has a major negative impact on TDF, leading to its destruction. These results highlight the importance of forest protection for both the conservation and restoration of TDF, which is a global priority (Miles *et al.,* 2006) as well as an urgent requirement in Mexico (Gordon and Newton, 2006a, b).

Case study in central Chile

This research was undertaken in a landscape in the central Chilean Mediterranean-climate region, focusing on an area that is currently being proposed for designation as a Biosphere Reserve. As a result of its high floristic endemism, the region is a global conservation priority, as illustrated by its inclusion in the Global 200 ecoregions (Olson and Dinerstein, 2000) and forms part of the Chilean Winter Rainfall-Valdivian Forests biodiversity hotspot (Myers *et al.*, 2000). LANDIS-II was used to simulate a range of different types of disturbance, including fire, stem cutting, herbivory and spread of invasive species, both individually and in different combinations. Analyses examined the impact of these different disturbance regimes on forest structure and composition.

Study area

The investigation was conducted in Quilpue, located in the Vth Region of Chile between 32º56'7" and 33º22'49"S, and 70º59'14" and 71º39'53"W. The study area is located in the coastal mountain range situated south of the city of Santiago, and covers an area of 170,897 ha, with an altitudinal range of 15 to 2129 m a.s.l. The entire study area has recently been proposed as a UNESCO Biosphere Reserve, but two national protected areas that have already been designated lie within its boundary, namely the Lago Peñuelas National Reserve (LP) and La Campana National Park (LC). The climate of the area is characterized by hot dry summers and cool wet winters with strong inter-annual variability owing to the El Niño – Southern Oscillation (ENSO) phenomenon. Mean annual temperature is 13.2°C and mean annual precipitation is 531 mm.

Vegetation of the area is characterized by a spatially heterogeneous mosaic of different types of dry forest, including some xerophytic plant species such as *Cactus* spp. Lower slopes are dominated by shrubland dominated by *Acacia caven*, whereas evergreen sclerophyllous forest occurs mostly on south-facing slopes and in sheltered ravines, where dominant tree species include *Cryptocarya alba, Lithraea caustica, Peumus boldus* and *Schinus latifolia.* Rapid population growth within the region in recent decades has led to profound changes in patterns of land use (Schultz *et al.*, 2010), resulting from logging, agricultural intensification (particularly vineyard and fruit cultivation, as well as corn and wheat cropping), and establishment of exotic tree plantations (principally *Eucalyptus globulus*). Grazing of livestock is also widespread in the study area. Native forests continue to be an important source of fuelwood and other non-timber forest products (NTFPs) for local communities.

Field survey

Forest structure and composition was assessed in 50 field plots, following the methods described by Newton (2007a). The plots each measured 25 x 20 m and were distributed throughout the study area using a stratified random approach, with strata defined according to five classes of aspect (north-, south-, east- and west-facing, and flat) and five classes of elevation (0–400 m, 4–800 m, 8–1200 m and >1200 m). Each individual tree was identified and its diameter at breast height (dbh) measured using a measuring tape. Wood samples of a representative subset of trees of each species were taken using an increment borer, for subsequent determination of age-diameter relationships in the laboratory.

Model parameterization and calibration

The inputs required by the model include a landtype or ecoregion map, which describes the ecological conditions influencing tree establishment, and an initial communities map, which describes the distribution and age of cohorts of each species at year 0. A land-cover map was produced based on classification of an unprocessed Landsat TM image (path 233, rows 83 and 84) acquired for November 2008 (Schulz *et al.*, 2010). The land-cover map produced featured eight land-cover classes (native forest, shrubland, cropland, urban, bare ground, water, pasture, plantation). The spatial resolution of the data was 90 m. All maps were produced and manipulated using ArcGIS 9.2 (© 1999–2006 ESRI Inc., Redlands, California) and Idrisi Andes (Clark Labs, Clark University, Worcester MA, USA), projected using WGS 1984 UTM Zone 19S.

A total of 20 ecoregions were defined according to the strata used in the field survey (see above), and mapped using an overlay of the land-cover map and a digital elevation model. The initial forest age structure and composition of each ecoregion was determined using the data obtained from the individual field plots. The establishment probability of each species in each ecoregion was also defined by referring to the field survey data. Those species found to be present in a particular ecoregion were assigned an establishment probability of 1, for the timestep of the simulation (10 years). Those species not present in a particular ecoregion, but present in another elevation class in the same aspect class, were accorded an establishment probability of 0.8. Lower probabilities were assigned to all species in north-facing ecoregions, and at higher elevations, reflecting the higher drought risk of such sites. In this way, the model was calibrated to simulate the influence of aspect and elevation on vegetation composition, to reflect the patterns observed in the field (Schulz *et al*., 2010).

The life history characteristics of the 21 native species encountered in the field survey plus *A. dealbata* were obtained from field observations, the scientific literature and a range of local experts (**Table 8.4**). In LANDIS-II, forest succession interacts with several spatial components (i.e. seed dispersal, fire and harvesting disturbances). In this investigation, the LANDIS-II Base Fire v2.1 extension was used to explore fire dynamics. LANDIS-II also requires a fire ecoregion map as an input. Analysis of spatial data for recent (1985–2007) fires in the area indicated that proximity to urban areas or roads is a major factor influencing ignition probability. Consequently, a map was produced by buffering around each road and urban infrastructure, according to four distance classes (0–1 km, 1–4 km, 4–8 km and >8 km). This map was then overlaid on the land-cover map, enabling 30 fire ecoregions to be defined, representing different combinations of land-cover type and buffer distance classes. Fire characteristics for each fire ecoregion were derived from the spatial database of fire history.

The harvest module of LANDIS-II (Base Harvest extension v1.2) was used to simulate the impacts of stem cutting and browsing animals. Both of these disturbances were modelled by the removal of specific cohorts, with browsing removing any cohorts <10 years old, and cutting removing all cohorts except the youngest. Harvesting impacts were distributed according to the ecoregion map, with each ecoregion considered as a separate stand type, nested within an individual management area. The management areas were defined according to the core and buffer areas of the two reserves (LP and LC), and the remaining part of the study area, giving five areas in total.

Scenarios

Once model calibration was completed, final values of the model parameters were used in a series of modelled scenarios. LANDIS simulations were conducted for 200 years. In each case, trees were prevented from establishing on water, cultivated land, forest plantations and urban land-cover types. Five replicated simulations (with varying random number seed) were performed for each disturbance scenario to explore the variability of model predictions. The time steps were set at 10 years for tree succession, 10 years for fire disturbance and 1 year for grazing. The list of ages for each species was therefore grouped into age cohorts as follows: ages 1 to 10 years (10), 11 to 20 years (20), 21 to 30 years (30), etc.

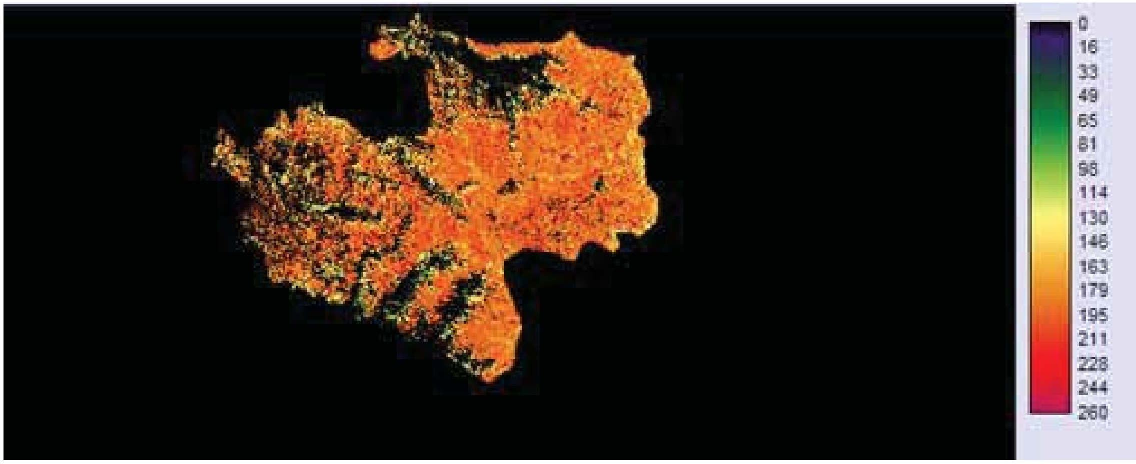
Seven different disturbance scenarios were simulated, all of which included the exotic species *A. dealbata*, except where stated: (1) no disturbance (excluding *A. dealbata*); (2) no disturbance (but including *A. dealbata*); (3) fire only; (4) fire and browsing, where browsing was applied randomly to 10% of the entire study area, including the protected areas; (5) fire and cutting, where all except the youngest trees were harvested, with the cutting applied to 0.1% of the protected areas (including both core and buffer areas), and 3% of unprotected areas; (6) fire, browsing and cutting, where cohorts <10 years old of target species were browsed, and cohorts >80 years old of target species (**Table 8.4**) were cut; both browsing and cutting were applied to 10% of the entire study area, including protected areas; (7) browsing and cutting implemented as for scenario 6, but without fire.

Results

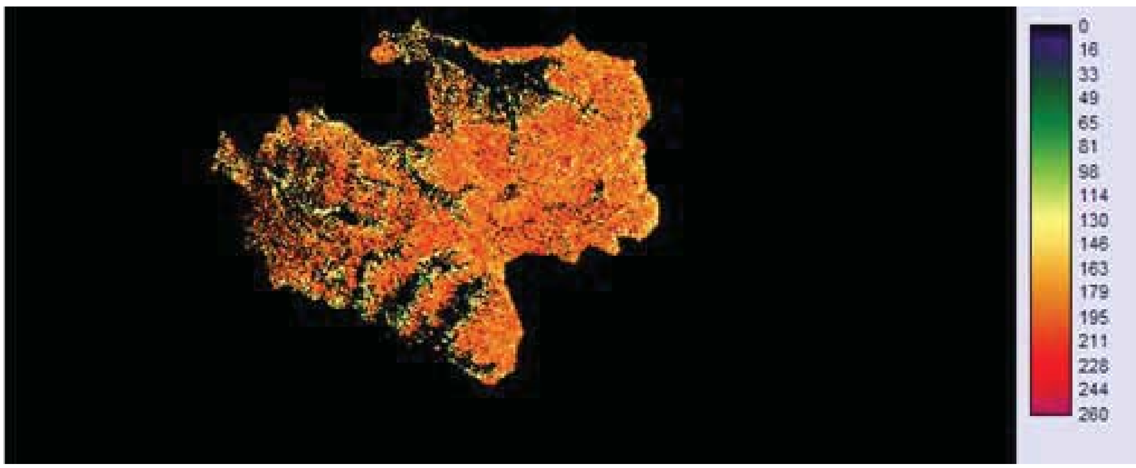
A brief summary of the results is provided here; full details are described by Newton *et al.* (2010) (see also **Box 8.2** for additional scenarios). Forest area increased rapidly during the first four decades in all scenarios, reaching values of around 121,000 ha in each case (**Fig. 8.5**). In scenario 1, forest area remained fairly stable thereafter. In the other scenarios, total forest area tended to decline slightly over time, particularly in scenario 6 (fire, browsing and cutting). However, the differences recorded between the scenarios in terms of total forest cover were slight, in each case accounting for <3% of the forest area present at the outset. More pronounced differences between scenarios were recorded in forest structure (**Fig. 8.5**). By the final timestep of the simulations (200 years), forest structure in the absence of disturbance (scenarios 1 and 2) was dominated by relatively 'old growth' forest stands, with 96% of forest area characterized by forest stands with trees >120 years old. All of the scenarios that included fire (scenarios 3, 4, 5 and 6) were characterized by high proportions of relatively young forest stands, each with 37–39% forest cover dominated by stands <40 years old. In scenarios with both browsing and cutting (6 and 7), either with or without fire, relatively old-growth forest stands (>120 years old) were virtually eliminated from the landscape, accounting for <11% of the forested area.

Contrasting results were also obtained between the different scenarios in patterns of species richness. In particular, species richness values tended to be lower in those scenarios without disturbance than those with burning, cutting and/or fire. For example, no forest areas were recorded in scenarios 1 and 2 with more than four tree species. In each of the other scenarios, substantial forest areas (>10,000 ha) were recorded with five or more species.

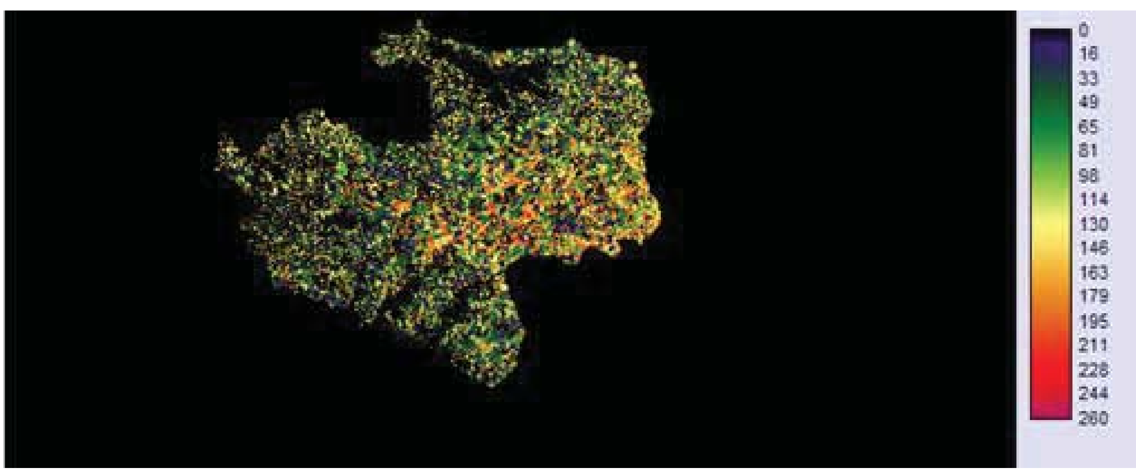
The dynamics of *Acacia dealbata* abundance differed from any of the native species considered. Cover values tended to increase rapidly in all scenarios during the first 40 years. Thereafter, in the absence of disturbance, the species declined rapidly such that it was largely eliminated from the landscape after 100 years. Declines were also recorded in scenarios 3 and 7, although in both cases values tended to remain stable after approximately 100 years. In the other three scenarios, after a period of stability, values increased continuously after 100 years. These results indicate that this introduced species is projected to spread only in the presence of fire when combined with browsing and/or cutting.



Scenario 1

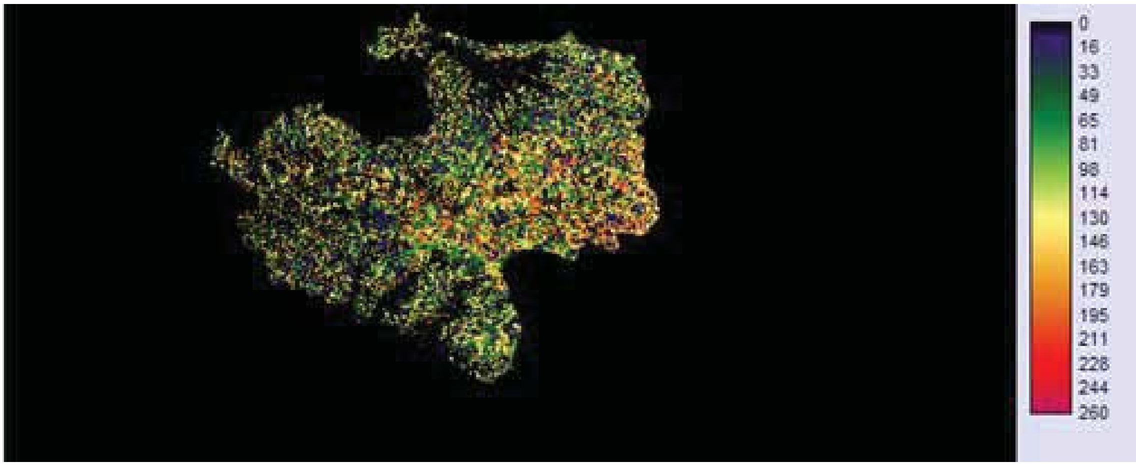


Scenario 2

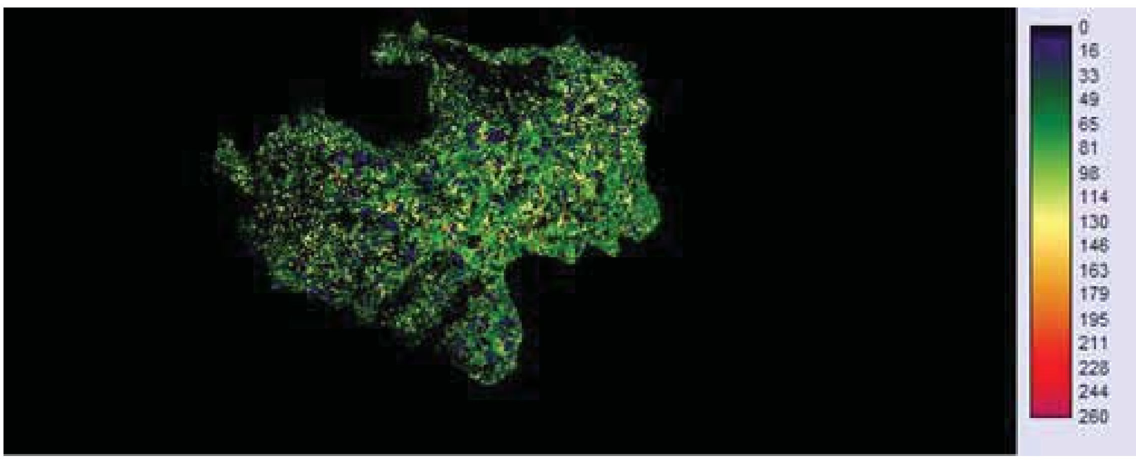


Scenario 3

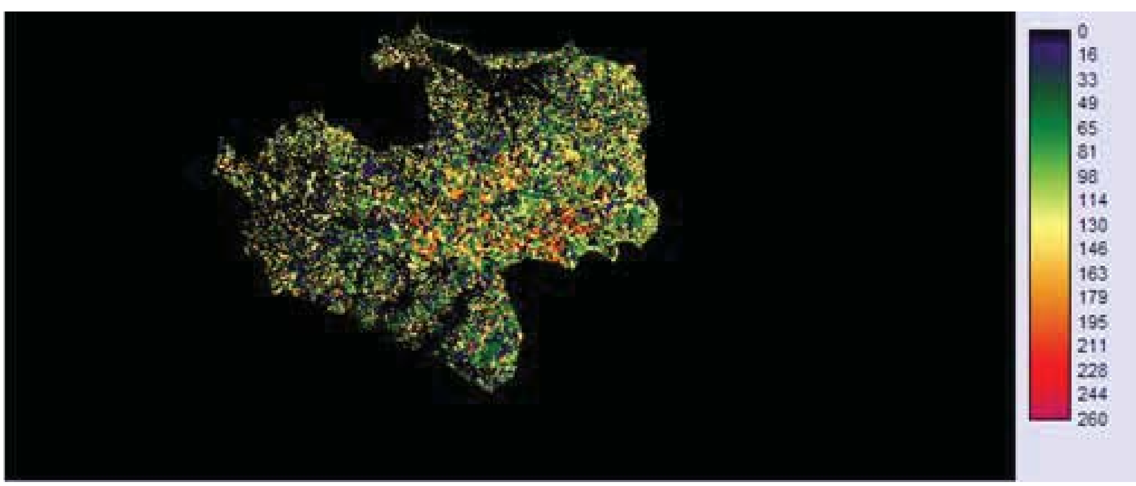
Figure 8.5 The above maps illustrate LANDIS-II model output, for the different disturbance scenarios conducted in Quilpue, Chile, over a period of 200 years. The different colours on the maps refer to different maximum ages of the forest stands. The maps provide illustration of projected forest landscape change in the future, under different disturbance regimes, with current land-cover as a starting point. Figure (a) scenario 1, (b) scenario 2, (c) scenario 3, (d) scenario 4, (e) scenario 5, (f) scenario 6, (g) scenario 7.



Scenario 4

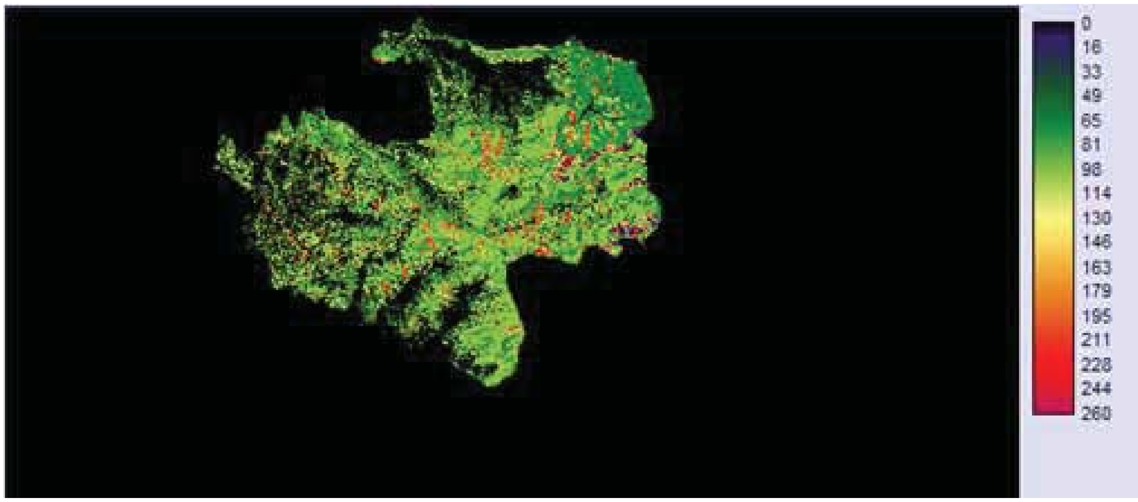


Scenario 5



Scenario 6

Figure 8.5 (cont.)



Scenario 7

Figure 8.5 (cont.)

Discussion

Results indicated that anthropogenic disturbance has a major impact on forest structure; when subjected to cutting and browsing, either with or without fire, old-growth forest was largely eliminated from the landscape. These results closely accord with those obtained from field surveys and analyses of remote sensing imagery conducted in the moist forests situated further south in Chile, where multiple types of anthropogenic disturbance are chronic and widespread (Newton, 2007b; Newton *et al.*, 2009a,b; Echeverria *et al.*, 2006; 2007; 2008; Wilson *et al.*, 2005).

The simulations also indicated that the characteristics of the disturbance regime had a major influence on patterns of species richness. In particular, species richness in the study landscape tended to be lower in scenarios with no disturbance than in those featuring fire, burning and/or cutting. These results are consistent with the 'intermediate disturbance hypothesis', which states that species richness is maximized at intermediate frequencies or intensities of disturbance (Connell, 1978). The hypothesis can be attributed to the role of disturbance in preventing the competitive exclusion of species.

The interactions between disturbance regime and ecological characteristics are most strikingly illustrated by the results obtained for *Acacia dealbata* in these simulations. In the absence of disturbance, this introduced species was largely eliminated from the landscape within a century, presumably as a result of competitive exclusion by native species. In the presence of disturbance, however, the species was able to maintain itself, or even spread through the landscape. This highlights the critical importance of effective management of the disturbance regime if invasive, alien species such as this are to be effectively controlled.

These results have a number of implications for conservation planning and management in this landscape. Currently, two national protected areas have been designated within the study area (LP and LC), and the entire area has recently been proposed as a UNESCO Biosphere Reserve. In this context, key planning challenges relate to how ecological restoration might be managed in different parts of the landscape, for example to increase connectivity between the two protected areas (see **Box 8.2**). While passive restoration might be encouraged by reduction of fire, cutting and browsing, the simulations presented here highlight the potential risks of preventing disturbance altogether, which might lead to a decline in species richness. However, if anthropogenic disturbance is to be tolerated or even encouraged in parts of this landscape to maintain species richness, then this will need to consider the potential risks of encouraging the spread of the invasive alien species, *Acacia dealbata*. Such potential conflicts highlight the need to explicitly consider tradeoffs during the management process, which could potentially be explored using the modelling approach described here.

Table 8.4 Species attribute table for the Quilpue study area. Long: longevity (years); Mat: age of sexual maturity (years); ShT: shade tolerance (1–5); FiT: fire tolerance (1–5); EffSD: effective seed dispersal distance (m); MaxSD: maximum seed dispersal distance (m); VRP: vegetative reproduction probability (0–1); MinVRP: minimum age of vegetative reproduction (years); MaxVRP: maximum age of vegetative reproduction (years); P-FiR: post-fire regeneration form (none, resprouting or serotiny). Dist refers to harvesting impacts; those species denoted 'C' were harvested in cutting scenarios, and those denoted 'B' were harvested in browsing scenarios.

Name	Long	Mat	ShT	FiT	EffSD	MaxSD	VRP	Min VRP	Max VRP	P-FiR	Dist
Acacia caven	150	20	1	3	1	150	1	10	80	re-sprout	B, C
Acacia dealbata	35	5	2	1	2	2000	1	10	35	re-sprout	
Aextoxicon punctatum	260	20	3	2	3	1000	1	10	200	re-sprout	B
Azara celastrina	100	20	3	1	10	100	0	0	0	none	
Azara dentata	100	20	3	1	10	100	1	10	80	re-sprout	
Beilschmiedia miersii	300	30	5	3	2	50	1	10	180	re-sprout	B
Cestrum parqui	50	25	2	3	2	50	0	0	0	none	B
Citronella mucronata	200	15	3	3	5	25	0	0	0	none	
Crinodendron patagua	180	20	3	1	4	100	0	0	0	none	B
Cryptocarya alba	150	20	2	4	8	1000	1	10	100	re-sprout	B, C
Drimys winteri	250	10	5	2	4	25	1	10	180	re-sprout	
Ephedra chilensis	100	20	2	3	8	100	1	10	80	re-sprout	
Lithraea caustica	200	20	3	3	8	1000	1	10	170	re-sprout	B, C
Lomatia hirsuta	100	20	4	2	5	100	1	10	70	re-sprout	
Luma apiculata	200	5	4	1	4	100	0	0	0	none	
Maytenus boaria	120	5	3	3	4	1000	1	10	80	re-sprout	B
Persea lingue	250	25	4	2	4	1000	1	10	200	re-sprout	
Peumus boldus	250	30	5	2	8	1000	1	10	200	re-sprout	B, C
Quillaja saponaria	200	25	4	4	4	100	1	10	180	re-sprout	B, C
Schinus latifolia	150	10	2	3	8	1000	1	10	100	re-sprout	B, C
Senna candolleana	150	15	1	1	1	150	1	10	100	re-sprout	B
Trevoa trinervis	150	20	3	1	4	50	1	10	90	re-sprout	

Box 8.2 Effects of fire on sclerophyllous forests in the Biosphere Reserve La Campana-Peñuelas in central Chile

A. Miranda, C. Echeverría, G. Bolados, E. Cantarello, A.C. Newton

Fire is one the main disturbance agents of the sclerophyllous forests in central Chile (Armesto and Gutierrez, 1978; Holmgren *et al*., 2000). Although some native plants can regrow after fire (Araya and Avila, 1981; Gómez-Gonzalez, 2009), the high frequency of fire in some areas has led to a considerable loss of forest (Miethke, 1993; Schulz *et al*., in press). The Valparaíso region concentrates the highest fire frequency in Chile. On average, 962 fires destroy 8600 ha per year (CONAF, 2009). Fire occurrence is closely related to human activities such as urban expansion and clearance for agriculture and livestock. The recently created Biosphere Reserve La Campana-Peñuelas aims to recover degraded zones by fire and livestock and convert commercial plantations of exotic species to native forests (CONAF, 2008).

In this study we assessed the effect of fire on the dynamics of the sclerophyllous forest in the Biosphere Reserve using LANDIS-II, base fire v2.1 (Sheller and Domingo, 2009). This version is able to simulate the spatial distribution of fire considering the ignition probability, fuel type, and fire rotation interval among others. Initial community maps included '*espinales*', an anthropogenic pseudo-savannah, and sclerophyllous forest formed by evergreen, hard-leaved tree species. A fire ecoregion map of 12 categories was built based on vegetation types (forest, espinal and grassland) and distance to urban areas (0–1000 m, 1000–4000 m, 4000–8000 m and greater than 8000 m).

The effect of fire on the spatial distribution and composition of tree species was analyzed for the following two scenarios: (i) scenario 1 as usual, in which the fire regime observed over the last 25 years is maintained. In this case, simulation parameters are calibrated for 4500 fires affecting 22,000 ha of native forest (CONAF, 2009); (ii): scenario 2 of landscape management, in which forest plantations of exotic species are replaced by native species that have been successful in forest restoration such us *Acacia caven, Maytenus boaria, Schinus latifolius* and *Quillaja saponaria*. Also, this scenario included the restoration in some specific riparian zones in order to enhance forest connectivity and the reduction of fire frequency and burnt area in 50%.

Under scenario 1 all the species analyzed exhibited a gradual decline in their abundance over the next 100 years (**Fig. 1**). *S. latifolius* and *Cryptocarya alba* showed a higher decline in abundance of 43% and 30% respectively, while *A. caven* and *Q. saponaria* decreased 5% and 2% respectively. Under scenario 2, the species more resistant to fire reached a higher regeneration rate than the forest loss rate by fire (**Fig. 2**). For instance, *A. caven* and *Lithrea caustica* increased 0.5% and 3% in abundance after 100 years respectively. On the other hand, the more vulnerable species to fire such as *S. latifolius* and *C. alba* decreased by 22% and 28% respectively. Although these two last species were favoured in abundance by a reduction in the fire regime parameters, *S. latifolius* exhibited a greater decrease in the loss of area, from 43% (scenario 1) to 22% (scenario 2).

Box 8.2 (cont.)

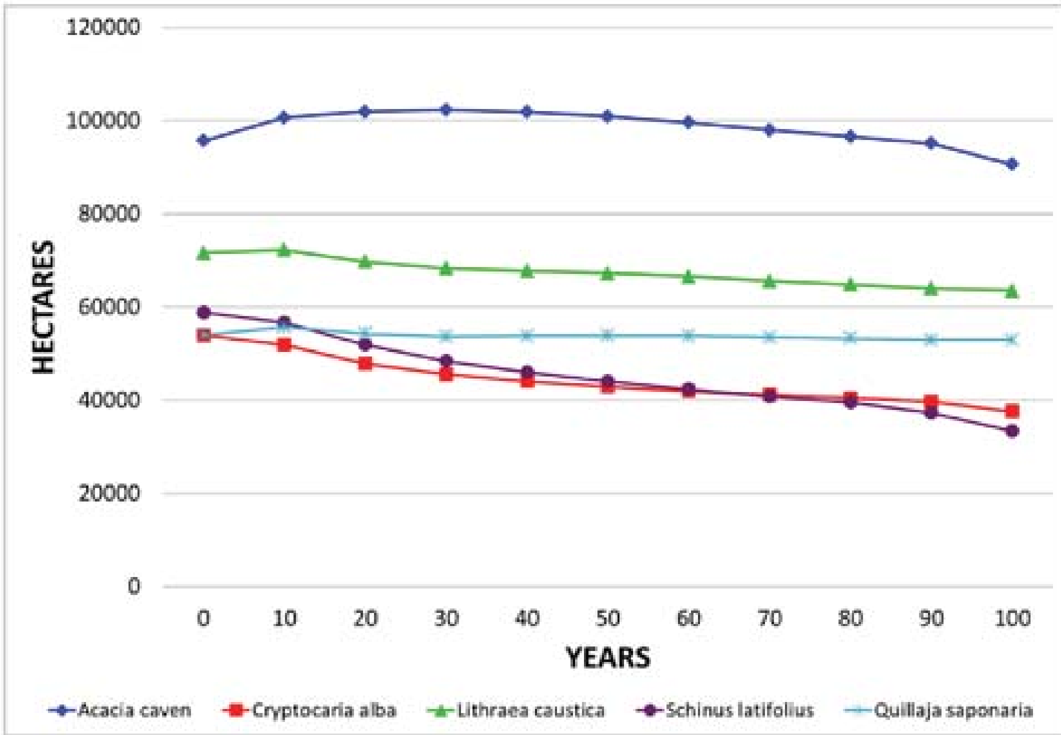


Figure 1 Abundance of the main species in the Biosphere Reserve La Campana-Peñuelas (ha) under scenario 1.

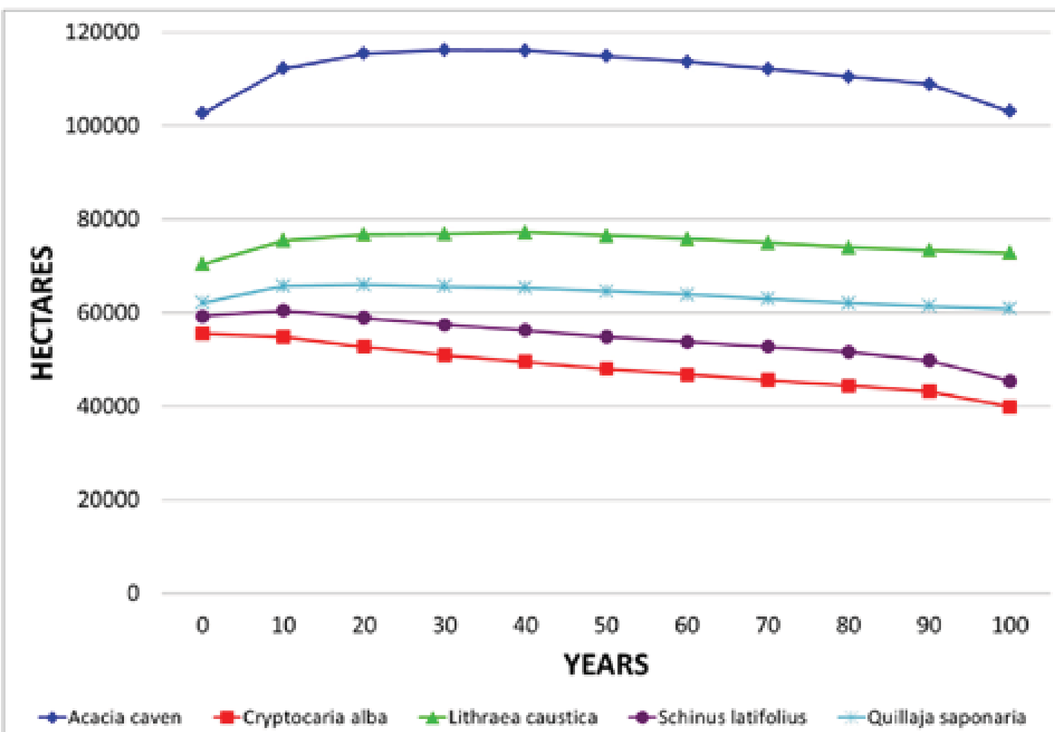


Figure 2 Abundance of the main species in the Biosphere Reserve La Campana-Peñuelas (ha) under scenario 2.

Box 8.2 (cont.)

Simulations revealed that the restoration actions along with a reduction in 50% in fire frequency and area caused a decline in the loss rate of some tree species in the Biosphere Reserve. Despite these actions, some species still continued to decrease over the next 100 years. On the other hand, fire-resistant species exhibited a slight increase in abundance in the landscape. Our results also showed that the selection of species used in the restoration areas was suitable to increase forestland in the study area. The species were able to survive and grow in the restoration area and to expand their propagules to the neighbouring sites.

It can be concluded that restoration actions are not the only measure to be implemented for the persistence of dryland forests in the Biosphere Reserve. It is also highly necessary to implement landscape planning and conservation actions that lead to a reduction in the occurrence of fire disturbances in the study area. The absence of this type of actions will probably lead to a gradual decline in some of the principal tree species in the Biosphere Reserve.

Case study in southern Argentina

This research was undertaken in the forest-steppe ecotone situated on the eastern slopes of the northern Patagonian Andes, Argentina. Climatically, the region is characterized by an abrupt precipitation gradient, which declines from west to east owing to the orographic effect of the Andes. As a result of droughts and the presence of both natural and anthropogenic ignition sources, fire is widespread in the region. Although the impacts of fire on forest stand structure and dynamics have been documented in detail by previous research (Kitzberger, 2002; Kitzberger and Veblen, 1997; Kitzberger *et al*., 1997, 2001, 2007; Mermoz *et al*., 2005; Veblen *et al*., 1999, 2008), the spatial dynamics of fire, and their interactions with other types of disturbance, have not been examined previously. Other forms of human disturbance affecting forests in the region include herbivory, as a result of extensive sheep, cattle and goat ranching. The objective of this study was to examine the potential for passive restoration of forest in the study area, under different disturbance regimes. This was addressed by using the LANDIS-II model to explore the following hypotheses, based on previous research undertaken in the area: (i) high fire frequency during aboriginal and European settlement eras has led to increased forest fragmentation and restriction of forest into fire-free rocky refugia, where low fuel loads have permitted the survival of isolated forest patches; (ii) the fire regime prevailing at the present time would permit forest recovery and expansion in the absence of herbivory, which is currently restricting forest distribution to rocky refugia.

Study area

The investigation was conducted in an area located between 40º54'37" and 41º15'20"S, and 70º41'4" and 71º24'1"W, on the eastern slopes of the northern Patagonia Andes. The area is situated between 737 and 2195 m a.s.l., and is 228,289 ha in extent. The western region of the study area comprises the east-central part of the Nahuel Huapi National Park and the remaining land is divided into large privately owned *estancias* (large farms or ranches). Topographically, from west to east, the area includes the Andean cordillera, the lower foothills of which are intersected by glacial lakes and valleys, and the Patagonian plains at approx. 700 m a.s.l. An abrupt west to east decrease in the amount of rainfall has created a unique forest-steppe ecotone. Within the study area, precipitation declines from approximately 1800 mm to less than 500 mm per year across this ecotone. The distribution of precipitation is highly seasonal, with approximately 60% falling during May to August. Mean annual temperature is approximately 8°C and mean monthly temperatures vary from 2–14°C.

Austrocedrus chilensis is the only tree species present in much of the study area, although it grows alongside *Nothofagus pumilio* in the wetter western area (above 1000 m a.s.l.) of the ecotone. With increasing aridity the understory becomes less dense as *Chusquea* is replaced by shrubs and small trees such as *Maytenus boaria* and *Lomatia hirsuta*. At the most easterly edge of the study area, *A. chilensis* stands become sparser until the only remaining trees are found associated with rocky outcrops. In these driest areas, trees are replaced by bunch grasses and low-growing shrubs such as *Discaria articulata* and *Mulinum spinosum*. Water balance is negative from October to March, creating conditions conducive to fire. Fire frequency prior to Caucasian settlement was extremely high, as the Native American population set fires to drive guanaco (*Lama guanicoe*) into open areas for hunting (Veblen *et al.*, 1999, 2008). Forest stands are now highly fragmented, and are also currently under threat from heavy grazing by livestock (principally cattle and sheep). In addition, introduced herbivores including European hares, rabbits and exotic deer are negatively impacting native arboreal vegetation.

Model parameterization and calibration

LANDIS-II requires parameters to be defined for each tree species included in the model, relating to a range of different ecological characteristics. Here, a single tree species (*Austrocedrus chilensis*) was incorporated in the simulations, reflecting its monodominance of ecotonal forests in the area. Species attributes were derived from the scientific literature, supported by field observations, as follows: maximum longevity, 500 years; age of sexual maturity, 20 years; shade tolerance, 2 (on a scale of 1–5); fire tolerance, 2 (on a scale of 1–5); effective seed dispersal distance, 15 m; maximum seed dispersal distance, 200 m; vegetative reproduction probability, 0; minimum age of vegetative reproduction, 0 years; maximum age of vegetative reproduction, 0 years; post-fire regeneration form, none.

A land-cover map was produced based on classification of an unprocessed Landsat TM image acquired for February 2003. Classification was achieved using field points to train the spectral signature of the selected land-cover classes in a supervised classification scheme, using a maximum likelihood algorithm. The map produced featured the following land-cover classes: wet forest, dry forest (*A. chilensis*), shrubland, wet grassland, dry grassland, bare ground, exotic plantations, urban, burned areas, and water bodies. The spatial resolution of the data was 28.5 m. In addition, a detailed distribution of *Austrocedrus chilensis* distribution and rock outcrops was produced by digitizing each outcrop from the high resolution orthophotos. The outcrops were considered as a separate land-cover type in the land-cover map. All maps were produced and manipulated using IDRISI Andes (Clark Labs, Clark University, Worcester MA, USA), projected using National Grid Argentina Faja1. LANDIS-II also requires a fire ecoregion map as an input. This was derived from the land-cover map, by assigning different fire characteristics to each land-cover type, based on available data. In this way, each land-cover type was modelled as a separate fire ecoregion. The harvest module of LANDIS-II (Base Harvest extension v1.2; Gustafson *et al.,* 2000) was used to simulate the impacts of browsing animals. This type of disturbance was modelled by the removal of a specific cohort (<10 years old). A map of potential forest distribution was also produced, to provide an indication of areas where forest could potentially be restored, and where forest might have been present prior to human disturbance. This map was derived from the analysis of current forest distribution in relation to climatic variables, using Mahalanobis typicalities implemented in IDRISI.

Scenarios

LANDIS simulations were conducted for 300 years. For all scenarios, water bodies, urban areas and wet forest were excluded from the simulations. The time steps were set at 10 years for tree succession, 10 years for fire disturbance and 1 year for grazing. The list of ages for *A. chilensis* was grouped into age cohorts as follows: ages 1 to 10 years (10), 21 to 30 years (20), 31 to 40 years (30), etc. A range of different scenarios were simulated but six of these are presented here by way of illustration: (1) no disturbance, (2) current fire regime, (3) historical fire regime, (7) harvesting and browsing only, at high intensity, without fire, (8) harvesting and browsing at low intensity, with current fire regime, (9) browsing at low intensity with historical fire regime.

Results

In all scenarios, forest cover increased continually over time (**Figs. 8.6**, **8.7**). The rate of increase in forest area was highest in scenario 1 (no disturbance), and lowest under scenario 9 (browsing and historical fire regime). Forest cover also increased relatively rapidly under the current fire regime, without grazing (scenario 2), but less rapidly under the historical fire regime (scenario 3). These results highlight the additive effects of browsing and fire in reducing the potential for increase in forest area, and also illustrate the potential impacts of different fire regimes on the rate and extent of forest recovery.

Discussion

The preliminary results presented here for this study area highlight the potential for forest recovery in southern Argentina, if the disturbance regime was managed appropriately. In all of the scenarios explored, an increase in forest area over time was observed, indicating that forest restoration might be achieved even under disturbance regimes featuring fire and browsing. However, rates of increase in forest area were found to be highest in situations without any form of disturbance. In each case, forest expanded from the remaining forest fragments, but at a rather low rate. The projections presented here suggest that many centuries would be required to re-establish forest cover throughout its former distribution, prior to human settlement. For this reason, practical restoration efforts may need to consider active restoration approaches (involving establishment of plantations of native tree species) in addition to passive approaches. Potentially, modelling approaches such as that presented here could be used to identify those locations within a landscape where passive restoration is likely to be effective or not over a given timescale. Locations where passive restoration is less likely to occur might usefully be targeted for active restoration. In addition, many target areas where passive restoration is feasible are currently being transformed into exotic pine plantations (Chapter 2). Modelling approaches such as the one presented here allow for a fine-tuned identification of areas of potential conflict between restoration of native dryland forests and other land uses.

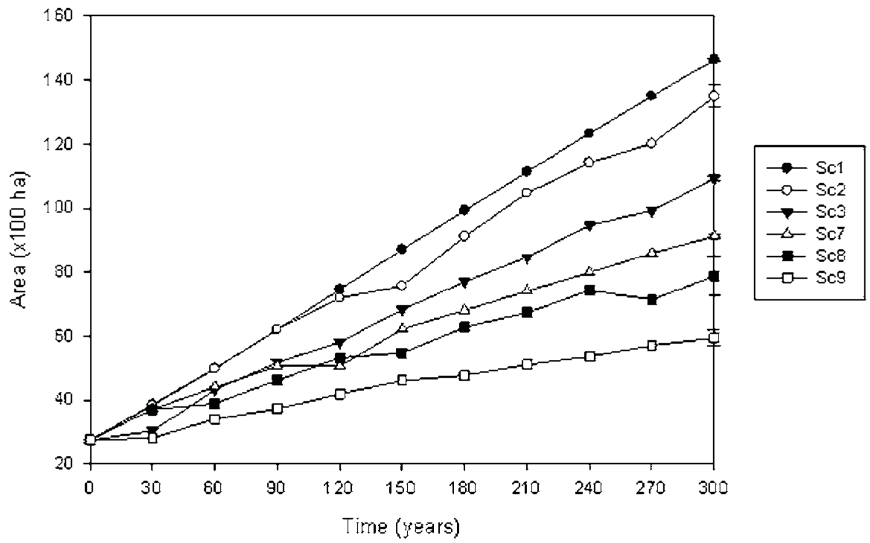
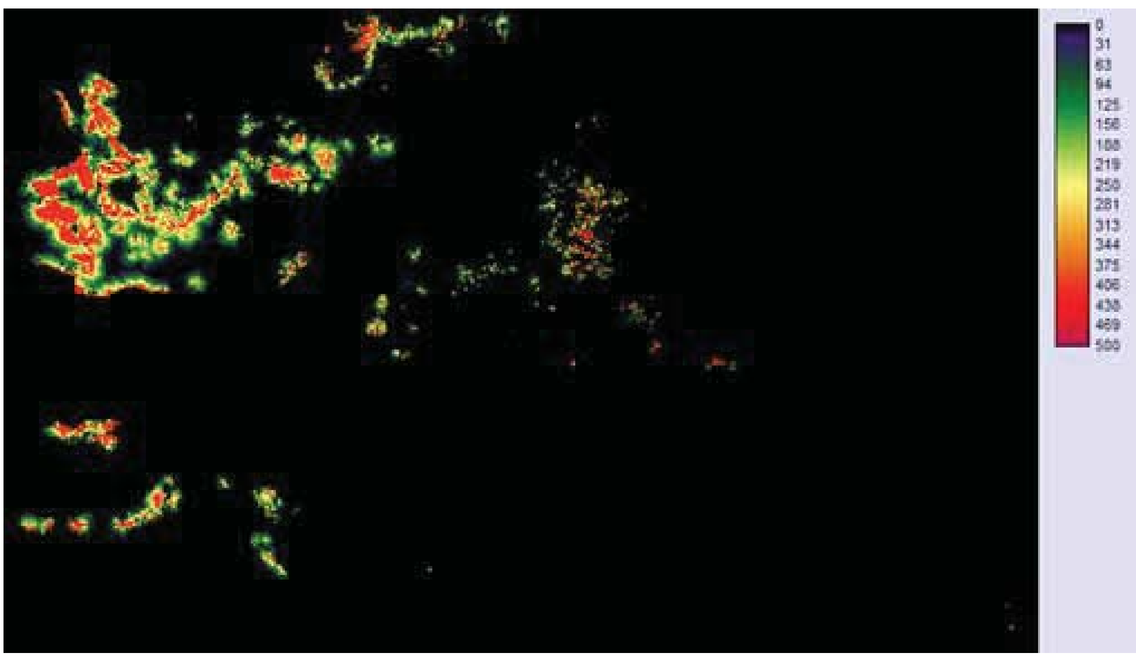
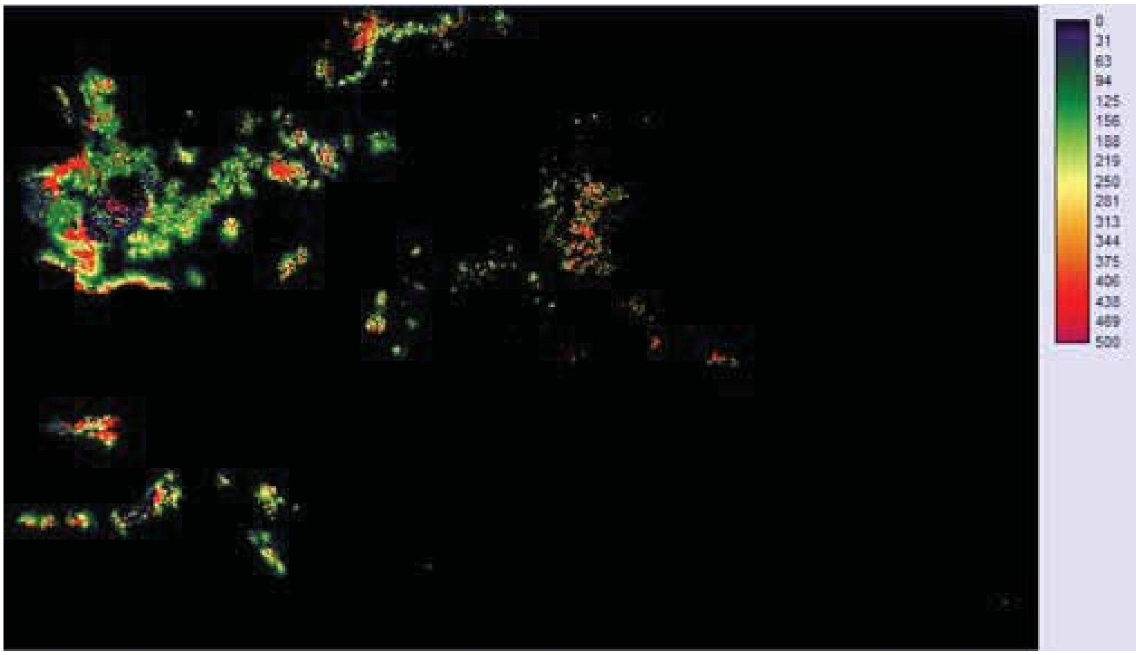


Figure 8.6 Changes in forest area in southern Argentina projected under different disturbance scenarios, by the LANDIS-II model. For details of scenarios, see text.

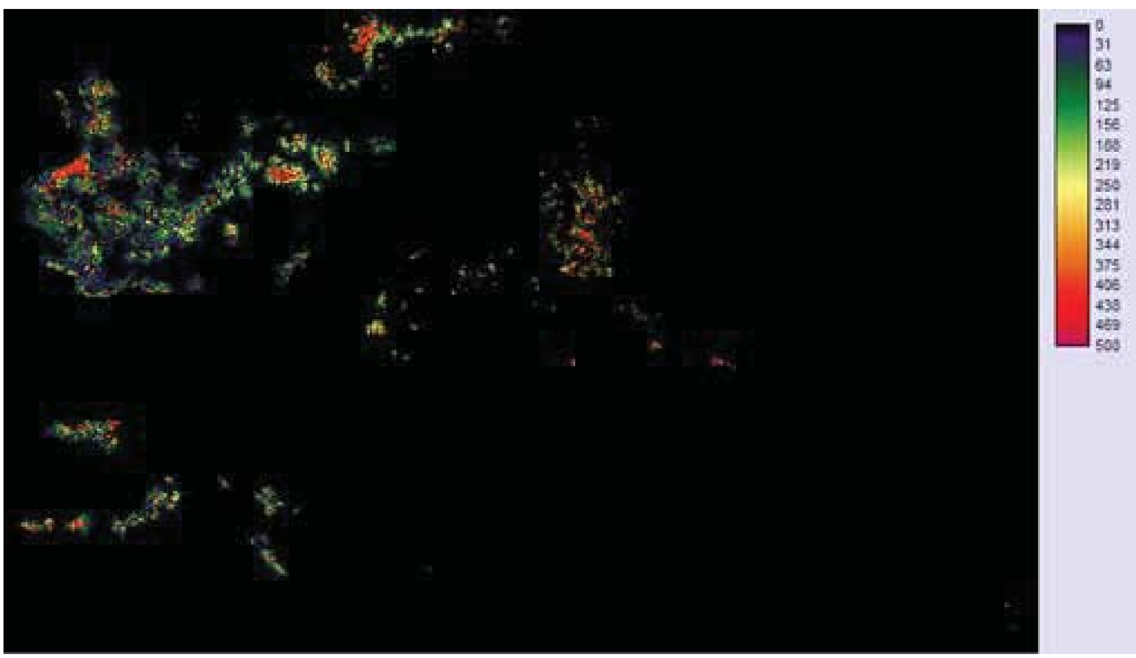


Scenario 1

Figure 8.7 The following maps illustrate LANDIS-II model output, for the different disturbance scenarios conducted in southern Argentina, over a period of 300 years. The maps provide an illustration of projected forest landscape change in the future, under different disturbance regimes, with current land-cover as a starting point. The different colours on the maps refer to different maximum ages of the forest stands. Figure (a) scenario 1, (b) scenario 2, (c) scenario 3, (d) scenario 7, (e) scenario 8, (f) scenario 9.

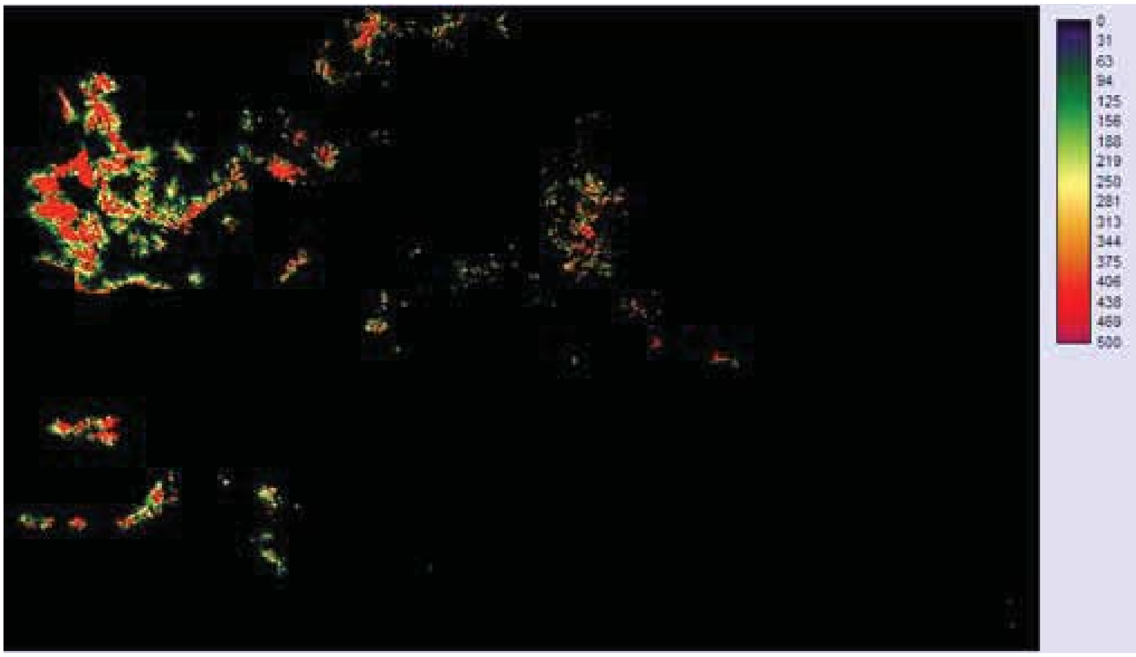


Scenario 2

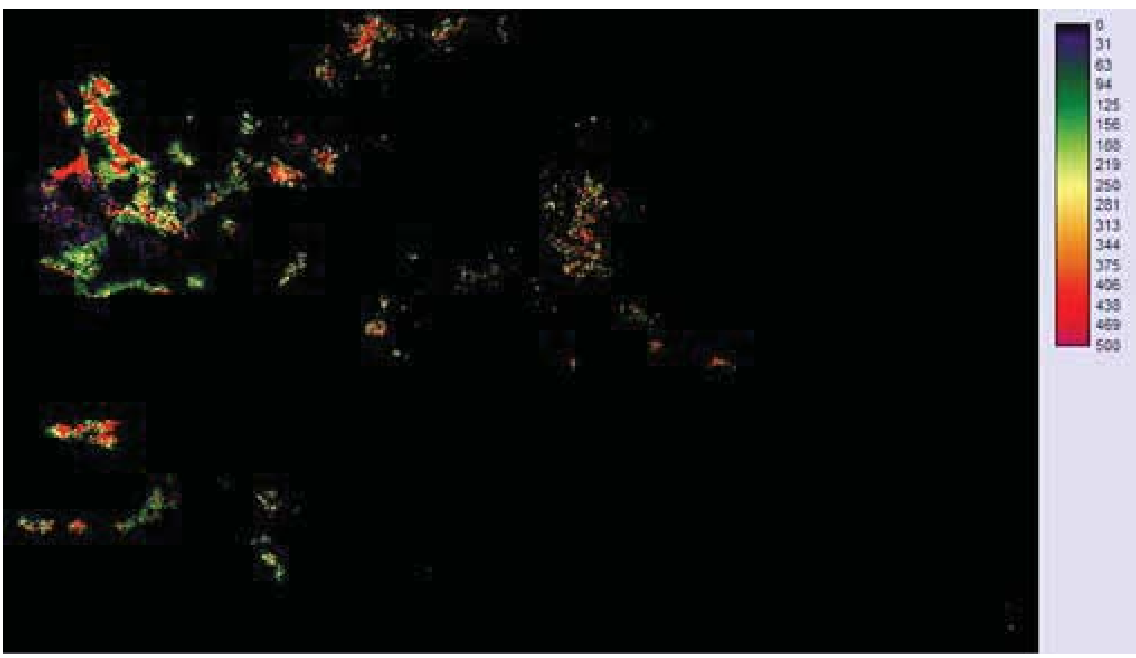


Scenario 3

Figure 8.7 (cont.)

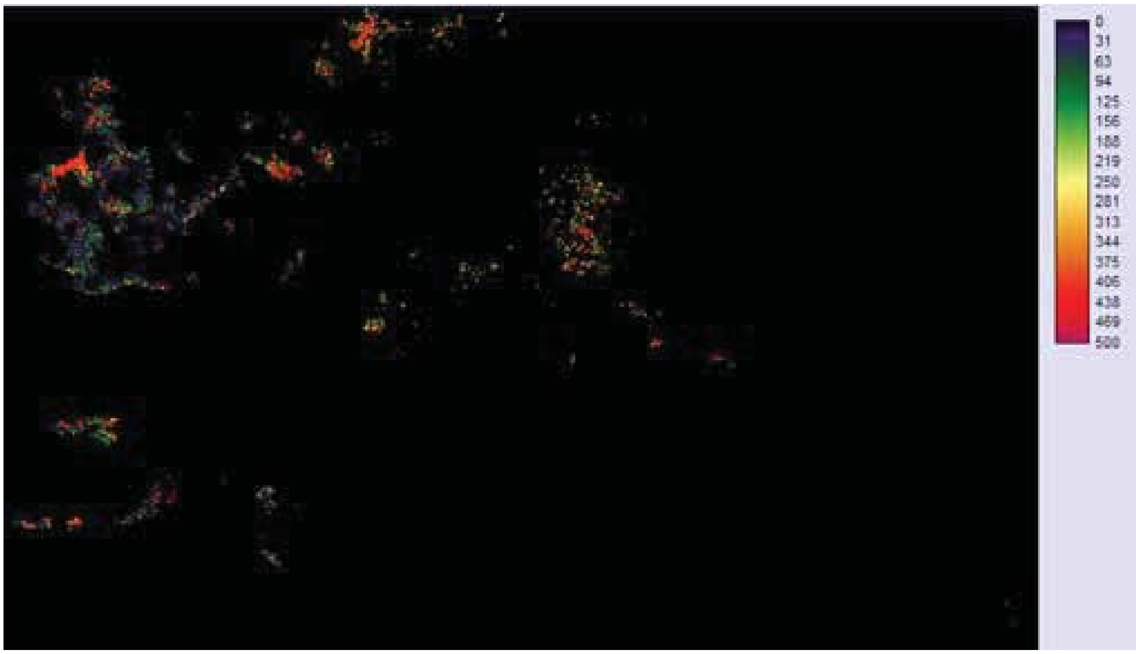


Scenario 7



Scenario 8

Figure 8.7 (cont.)



Scenario 9

Figure 8.7 (cont.)

Discussion

As noted by Bugmann (2001), the value of forest models does not necessarily lie in their ability to make accurate predictions, but in their ability to help understanding of processes and patterns by allowing exploration of the consequences of a set of explicitly stated assumptions. The outputs can be viewed as hypotheses that require further testing. Although ecological models have been employed to explore forest dynamics in a wide range of different locations, they have rarely been used to support forest restoration planning and practice (Newton, 2007a). This partly reflects a traditional focus on modelling the dynamics of individual forest stands rather than landscapes. It is now widely appreciated that understanding the spatial dynamics of forested landscapes is essential to their effective management and conservation (Newton, 2007b; Gardner *et al.*, 2009; Lindenmeyer and Franklin, 2002). In this context, recent progress in developing spatially explicit forest modelling approaches is of particular value, enabling spatial processes such as dispersal and colonization to be incorporated. As illustrated here, the LANDIS-II model provides a valuable tool for exploring the dynamic patterns of forested landscapes under different disturbance regimes, and can provide insights into the potential for restoration of forest landscapes. A further advantage of using a modelling approach such as LANDIS-II is that it can help identify knowledge gaps, and provide a framework for focusing future research efforts.

One of the main limitations common to all process-based models of forest dynamics, including LANDIS-II, is the difficulty of obtaining rigorous model validation, owing to the general lack of long-term data describing the ecological behaviour of forests (Newton, 2007a; Shugart, 1984; Shugart and West, 1980). The current examples are no exception; relatively little ecological research has been undertaken previously in the forests described here. This lack of information also hinders model parameterization. In particular, information is lacking on the dispersal ability of different tree species, a process that has a major influence on model outputs. Other areas of uncertainty include the precise impacts of herbivory and fire on the mortality of different age cohorts of trees, and variation in the establishment probability of different tree species across the landscape. As a consequence, the results presented should be viewed as highly tentative. However, as demonstrated here, the use of high resolution remote sensing imagery can be of value providing detailed information about forest structure and distribution, especially in areas that are relatively inaccessible (for example in Chiapas). The integration of such imagery with spatial modelling approaches offers a powerful set of tools with which to explore the sustainable management of forest landscapes (Newton *et al.,* 2009a,b). Further elements of an emerging toolkit could potentially include models that enable the potential impacts of climate change to be explored (**Boxes 8.3** and **8.4**).

Conclusions

The results presented here highlight the value of spatially explicit modelling tools for exploring the potential for restoration of forest landscapes. Specifically, the modelling approach used (LANDIS-II) enabled projections to be made regarding the pattern of regeneration and spread of native forest under different anthropogenic disturbance regimes, providing an insight into the potential for passive restoration approaches. The model also highlighted interactions between different forms of disturbance and their impacts on restoration processes, an area in which information is currently lacking. For example, modelling scenarios conducted in Chile indicated how spread of the invasive exotic species *Acacia dealbata* is dependent on other forms of disturbance such as grazing and fire. These examples demonstrate how spatial models can inform approaches to forest landscape restoration, by indicating those locations within a landscape where particular restoration approaches are most likely to be successful. In addition, spatially explicit modelling tools provide a means of visualizing the potential impact of restoration actions at the scale of entire landscapes.

Box 8.3 Effects of climate change on subtropical forests of South America

S. Pacheco, L. R. Malizia, L. Cayuela

Premontane forests in northern Argentina and southern Bolivia represent the lowest vegetation belt of the Yungas or subtropical montane forests. They are a conservation priority as they play a key ecological role (Brown and Malizia, 2004; Brown, 2009). These forests have been subjected to a long history of use (Malizia *et al.,* 2009; Brown *et al.,* 2006; Fundación ProYungas, 2007) and climatic variation (Prado, 1995). The objective of this research was to determine the future distribution of the premontane forest as a response to climate change, and to analyze the consequences of this distributional shift for its conservation and restoration.

To determine changes in forest distribution we developed statistical models (Scout *et al*., 2002) using Maxent (Phillips *et al*., 2006; Phyllips and Dudik, 2008). Location points indicating forest presence were obtained from: (1) the Subtropical Network of Permanent Plots (Fundación ProYungas, 2007; Blundo and Malizia, 2009), and (2) rapid assessment of tree species using 0.1 ha circular plots. The current and future variables of the CCM3 scenario, for the end of the century (Govindasamy *et al*., 2003) were obtained from the WorldClim database (Hijmans, 2005).

Box 8.3 (cont.)

The variables used were: mean annual precipitation, precipitation of driest month, temperature seasonality, and maximum temperature of warmest month. To obtain more accurate models we calibrated the mean annual precipitation data with Bianchi's local precipitation model (Bianchi *et al.,* 2008).

Once fitted, present and future distribution models were overlaid to define stable forest areas and areas of potential change (expansion and retraction). We defined as stable forest areas those which currently correspond to premontane forests and are likely to continue as forests in the future. The expansion areas are those that do not correspond to premontane forests at the present time but, as a consequence of climate change, could be colonized by species typical of this environment. Retraction areas are those areas of premontane forest that will shift to other potential vegetation types in the future.

The results obtained indicate that according to the internal validation executed by Maxent, the current distribution model of premontane forests has a high overall accuracy (AUC 0.95 ± 0.013). Ten percent of the original area of premontane forest is now transformed into agricultural land, mainly concentrated in flat areas below a 5% slope. The model of future distribution predicts a 53% decrease in cover as compared to the current area occupied by premontane forest. Of the future potential area occupied by this forest, approximately 30% is represented by the original cover and the remaining corresponds to areas of potential expansion owing to suitable changes in climatic conditions. The future climate scenario shows increases of *ca.* 1 °C in temperature and 80 mm in mean annual precipitation. According to our models, such changes will trigger an upwards altitudinal shift of premontane forests of about 300 m a.s.l. in the mountains. The largest retraction of the current forest area will occur in the northeastern range of its distribution, whereas stable areas will be concentrated mostly along the western sector and the southeastern range of its distribution. The latter is also a potential expansion area under climate change (**Fig. 1**).

Currently, 8% of the premontane forest is included in the protected areas of Acambuco, Piarfon, Pintascayo, Pizarro (Salta), and Calilegua (Jujuy) (**Fig. 1**). Under a climate change scenario, only 50% of this surface will remain as premontane forest. Premontane forests are projected to disappear in Acambuco, Piarfon and Pintascayo, and remain in Calilegua, Pizarro, Lancitas (Jujuy), and the Biosphere Reserve of the Yungas (**Fig. 1**).

To the best of our knowledge, our study is the first to evaluate the potential effects of climate change on premontane forest distribution in Argentina. Our results show that changes in climate conditions will markedly affect the distribution of premontane forest. A decrease in the area occupied and an upward migration of premontane forests is likely to occur in response to climate change, mostly as a result of an increase in temperature. Areas of potential expansion are currently covered by montane forest, which constitutes the immediate vegetation belt above premontane forests. The stable areas are mainly located in the western range of its distribution, which include the Yungas Biosphere Reserve. The northeastern range of its distribution, on the contrary, is predicted to suffer a sharp contraction. In relation to these shifts, protected areas located in the west and southeast of the study region are more likely to preserve premontane areas in the future, whereas the northeastern protected areas are predicted to lose their premontane forest in the future. The latter areas are likely to be colonized by species from the chaco forest and by premontane species tolerant of warmer weather conditions. Owing to the spatial and functional connection between the Chaco forest and the premontane forest, the current system of protected areas would probably help maintain their connection.

Box 8.3 (cont.)

Figure 1 Stable areas and areas of change (expansion, retraction) for premontane forests based on current and future potential climatically-based distribution models using Maxent. Premontane forests in retraction and stable premontane forests make up the current area occupied by premontane forests. Premontane forests in expansion and stable premontane forests make up the future distribution of premontane forests. References for protected areas: (1) Baritú, (2) Pintascayo, (3) Acambuco, (4) Piarfon, (5) Calilegua, (6) Serranias de Zapla, (7) Lancitas, (8) Pizarro, (9) El Rey.

Box 8.4 Effects of climate change on dryland ecosystems of central Chile

G. Henriquez Tapia, S. Pacheco, C. Echeverría

Worldwide, dryland ecosystems have been rapidly degrading. Dryland systems include all subhumid, arid, semi-arid and hyper-arid regions of the world where water is the main limiting factor of primary production and, consequently, other ecosystem services and human well being. Drylands occupy about 41% of the Earth's land area, comprising a high biodiversity, and providing a home to about two billion people. The persistence of drylands as a consequence of overexploitation is highly uncertain, especially now climate changes make predictions on future ecosystem functioning highly speculative. Generally, climate changes are thought to be accompanied by stronger and more frequent weather extremes (Easterling *et al.,* 2009; Meehl *et al.,* 2000). Among the species associated with this forest type are *Acacia caven, Beilschmiedia miersii, Colliguaja odonifera, Crinodendron patagua, Cryptocarya alba, Drimys winteri, caustic Lithraea, Maytenus boaria, Myrceugenia exsucca, Peumus boldus* and *Quillaja saponaria* (Donoso, 1995). Central Chile has been considered as one of the 25 high-priority areas of biodiversity (Hechenleintner *et al*., 2005) and one of the 34 biodiversity hotspots at a global scale.

In this work we modelled the current and future potential distribution of three tree species of the dryland forest in central Chile. We compared changes in their distributions using a scenario of climate change and the software MAXENT. All the input data were generated into a GIS using ARC GIS 9.3. The three species selected were *Acacia caven*, *Quillaja saponaria* and *Cryptocarya alba*. These species are socially important for local people as they are regularly used as a non-timber forest product (NTFP). Also, these species are present in one of the parts in Chile that will be severely impacted by climate change. Data of occurrence were obtained from the National vegetation mapping in Chile (known as Catastro de Bosque nativo, CONAF *et al*., 1999).

Climatic variables were obtained from the regional pattern PRECIS (Regional Providing Climates for Impact Studies), which was developed by the Hadley Centre of the United Kingdom and has a horizontal resolution of 25 km^2. PRECIS includes (i) current climatic variables for 1961–1990, and (ii) two models of future climate for the period 2071–2100: one moderate considering the scenario MESSRS B2 of the IPCC and another severe considering the scenario MESSRS A2 of the IPCC. In our work, we used the models of potential distribution and that of climatic change MESSRS B2.

The models of potential distribution, for the three species, presented a high general efficiency. The value of AUC for the pattern developed in the current scenario was 0.904 (0.022), and for the scenario B2 was 0.917 (0.019). They can therefore be considered as good current models of distribution to generate hypothesis for change in scenarios of climatic variation.

The models generated for the potential distribution of the three species are very similar to the current distribution of these species. Although the climate variables used in this work corresponded to earlier years (1961–1990) than those used for the occurrence of the species (1994–1997), the models of potential distribution derived from the climate variables appear to be suitable.

Our results revealed that in a future scenario, changes would in the distributions of the modelled species. In *A. caven* a reduction in the total area was observed without changes in the distributional range. In the case of *Q. saponaria*, projections indicate a reduction in the northern part of its range and for *C. alba*, a decline is projected in the northern and southern parts of its distribution (see **Figs. 1–3**).

Box 8.4 (cont.)

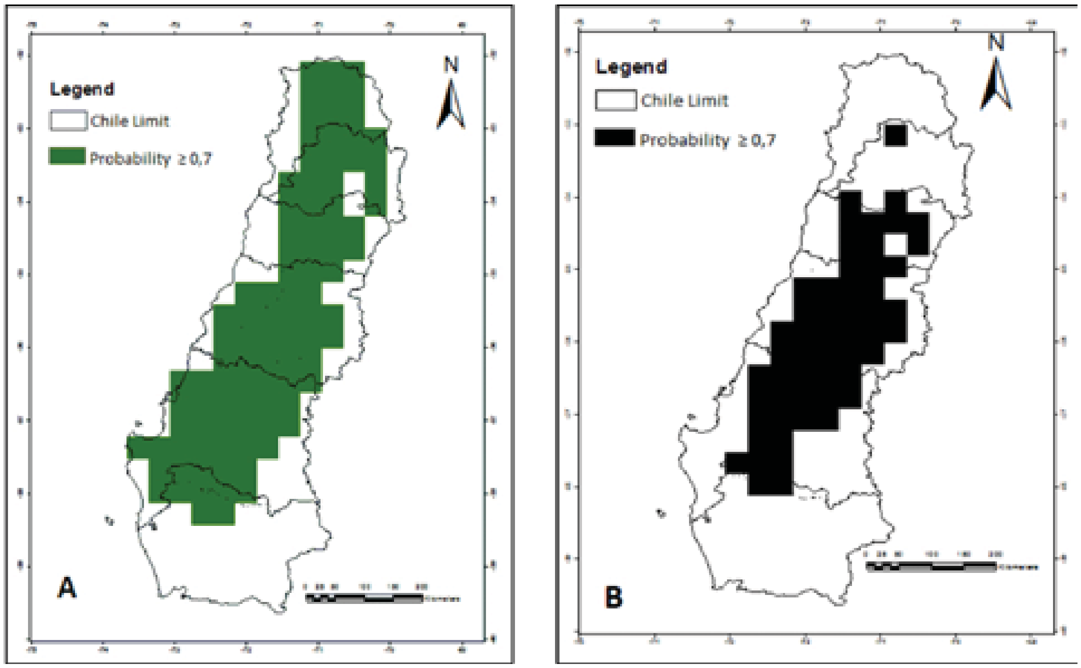


Figure 1. Maps of current potential distribution of *Quillaja saponaria* for a probability of occurrence higher than 0.7 (A) and potential distribution using B2 model of climate change (B).

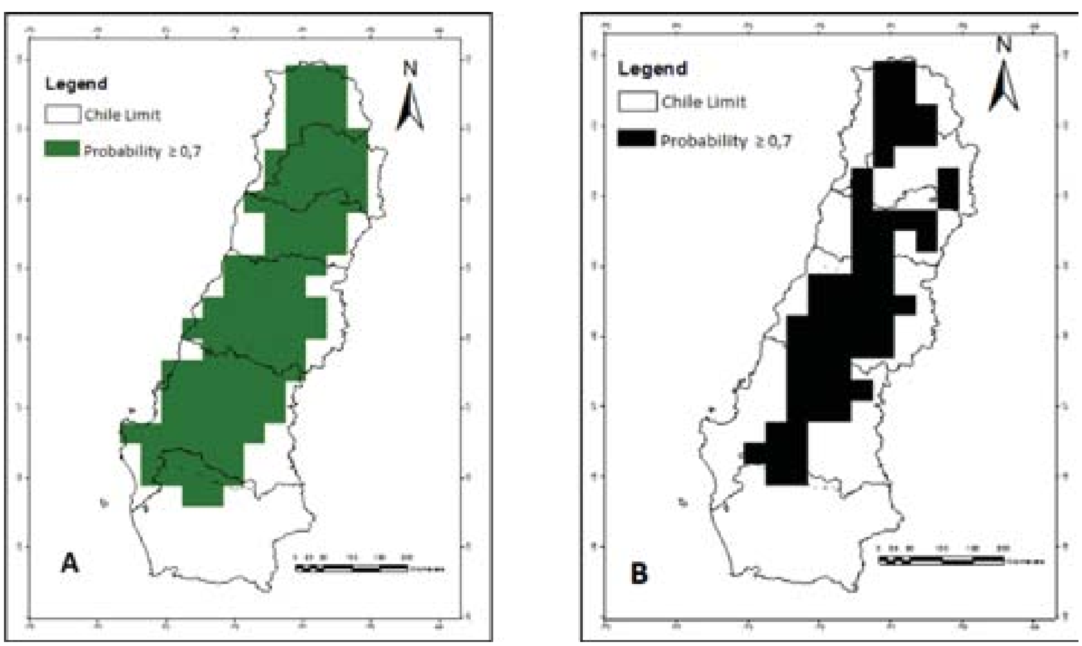


Figure 2 Maps of current potential distribution of *Acacia caven* for a probability of occurrence higher than 0.7 (A) and potential distribution using B2 model of climate change (B).

Box 8.4 (cont.)

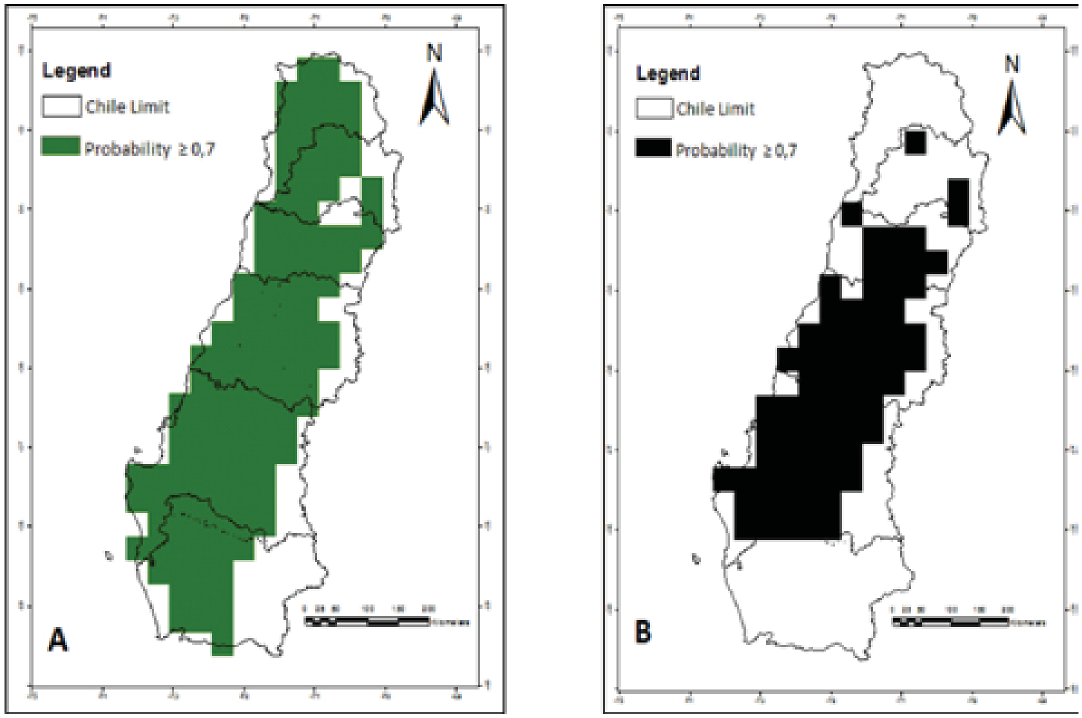
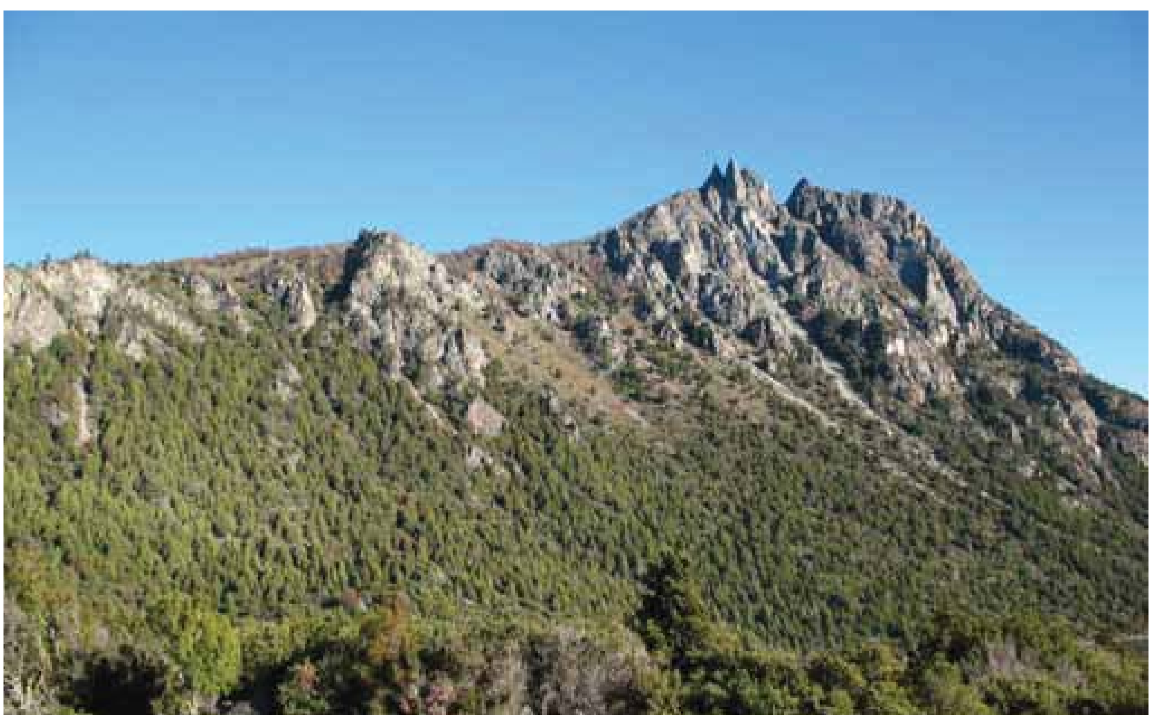


Figure 3 Maps of current potential distribution of *Criptocarya alba* for a probability of occurrence higher than 0.7 (A) and potential distribution using B2 model of climate change (B).



Austrocedrus chilensis **forest towards the western edge of its distribution in Nahuel Huapi, Argentina. Photo: A.C. Newton**

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9 IDENTIFYING PRIORITY AREAS FOR DRYLAND FOREST RESTORATION

D. Geneletti, F. Orsi, E. Ianni, A.C. Newton

Introduction

An urgent question in nature conservation is: where to act first? This is primarily related to concerns of an economic kind: financial resources are limited, hence conservation efforts should focus on areas where interventions will produce the greatest benefits. The prioritization problem has been addressed in a variety of ways (Mittermeier *et al.*, 1998; Roberts *et al.*, 2002). For example, a biodiversity hotspot is defined as an area with exceptional concentration of endemic species and with high rates of habitat loss, and can be seen as a priority for conserving the most species at the least cost (Myers *et al.*, 2000). Alternatively, species richness, endemism, unusual ecological or evolutionary phenomena and habitat rarity have been used at a global scale to identify ecoregions that should be accorded priority for conservation (Olson and Dinerstein, 2002). Research in conservation priority-setting has primarily focused on the design of protected area networks, which may be informed by analysis of the relative vulnerability of different areas to environmental pressures or threats (Wilson *et al.*, 2005).

Relatively little attention has been given to priority-setting in the specific context of forest restoration. In this context, the identification of priority areas depends upon the objectives of the intervention, which are often multiple and different in nature: enhancing biodiversity, providing local communities with financial and livelihoods benefits, etc. (Lamb and Gilmour, 2003; Mansourian *et al.*, 2005). Different objectives may result in the selection of different sites, as well as different restoration actions. Furthermore, selecting priorities requires, prior to the comparison of possible sites, their actual identification and design. These issues call for the use of methods able to integrate different types of variables, with varying levels of spatial accuracy, and to make the tradeoffs involved in the decision explicit. This chapter examines such methods, with specific reference to spatial multicriteria evaluation techniques, and explores their practical application to planning the restoration of dryland forest landscapes.

Multicriteria evaluation (MCE)

Multicriteria evaluation (MCE) techniques support the solution of a decision problem by evaluating the alternatives from different perspectives and by analyzing their robustness with respect to uncertainty. A characteristic feature of multicriteria approaches is that the evaluation is based on a number of explicitly formulated criteria, i.e., 'standard of judging', that provide indications of the performance of the alternatives with respect to a number of objectives. Such criteria, which are typically represented by considerable mutual difference in nature, are expressed by appropriate units of measurement. The nature of MCE makes it particularly suitable for environmental and natural resources decision-making. Such a type of decision problems involves multiple objectives and multiple criteria, which are typically non-commensurable and often conflicting.

A short description follows of the typical operational steps required to carry out an MCE to support decision problems. Complete overviews, as well as examples of applications of MCE for environmental management and land-use planning, can be found in Beinat and Nijkamp (1998). The starting point is the setting-up of an evaluation matrix, which contains the possible alternatives and the criteria against which they have to be evaluated. In the case of decision problems related to the prioritization of restoration interventions, the alternatives are typically represented by different areas or sites. The criterion scores consist of raw measurements expressed by different scales or units (monetary units, physical units, etc.). In order to be relatable to the degree of 'desirability' of the alternatives under analysis, such scores need to be transformed from their original units into a value scale. This is the role of the value assessment, through which the criterion scores lose their dimension and become an expression of the achievement of the evaluation objectives. This operation is performed by generating a value function, i.e. a curve that expresses the relationship between the criterion scores and the correspondent value scores (Beinat, 1997; Geneletti, 2005).

The different evaluation criteria are usually characterized by different importance levels, which need to be included in the evaluation. This is obtained by assigning a weight to each criterion. A weight can be defined as a value assigned to a criterion that indicates its importance relative to the other criteria under consideration (Malczewski, 1999). A survey of the methods developed to support the weight assignment can be found in Herwijnen (1999). Once the weights are assigned to each criterion, the aggregation can be performed. This is done by using a decision rule that dictates how best to order the alternatives, on the basis of the data on the alternatives (criterion scores), and on the preferences of the decision makers (criterion assessment and weights). The most widely used decision rule is the weighted linear combination. An overall score is calculated for each alternative by first multiplying the valued criterion scores by their appropriate weight, and then adding together the weighted scores for all criteria. Another popular method is 'concordance analysis' (Roy, 1985), which assesses the ranking by pairwise comparison of the alternatives. The last step in the procedure is represented by the sensitivity analysis. This is aimed at determining the robustness of the ranking with respect to the uncertainties in the assigned weights, value functions and scores, as well as to changes in the aggregation method (see examples in Geneletti *et al.*, 2003). The information available to decision makers is often uncertain and imprecise, owing to measurement and conceptual errors. Sensitivity analysis considers how, and how much, such errors affect the final result of the evaluation.

Spatial MCE to identify forest restoration priorities

The selection of forest restoration priorities is a complex land-use planning problem involving the collection and processing of information that relates to environmental, socioeconomic, as well as operational aspects. The spatial nature of the problem makes the use of Geographic Information Systems (GIS) necessary to easily manage geo-referenced data. MCE in a GIS environment (or spatial multicriteria evaluation) is a procedure to identify and compare solutions to a spatial problem, based on the combination of multiple factors that can be, at least partially, represented by maps (Malczewski, 1999; Geneletti, 2010). This approach takes advantage of both the capability of GIS to manage and process spatial information, and the flexibility of MCE to combine factual information (e.g. forest type, slope, infrastructure) with value-based information (e.g. experts and stakeholders' opinion, participatory surveys).

Ejido **Los Angeles , La Sepultura Biosphere Reserve, Chiapas, Mexico. Photo: N. E. Taylor-Aquino**

Milpa **production, La Sepultura Biosphere Reserve, Chiapas, Mexico. Photo: N. E. Taylor-Aquino**

Taking into account both technical elements and people's values and perceptions is essential to build consensus around a decision, to reduce conflicts, and consequently to pave the way to successful forest restoration interventions.

MCE and GIS can be coupled to provide spatial decision support, as shown in a number of applications related to nature conservation, environmental planning and forest management (Store and Kangas, 2001; Ceballos-Silva and Lopez-Blanco, 2003). GIS-based MCE has been specifically applied to prioritization studies: Bojórquez-Tapia *et al.* (2004) considered conservation priorities to design a National Park in Mexico; Geneletti (2004) ranked forest remnants according to their priority for conservation; Cipollini *et al.* (2005) modelled expert knowledge to prioritize the management of limestone prairies; Marjokorpi and Otsamo (2006) proposed a methodology to find rehabilitation priorities at the landscape scale.

The following sections explore some of the most critical issues related to the application of spatial MCE for forest restoration priority selection, namely:

- the identification of suitable criteria and indicators to guide the prioritization process;
- the involvement of experts and stakeholders, and the inclusion of their perspectives and values in the evaluation;
- the development and application of appropriate techniques to spatially aggregate criteria, to identify restoration options, and to perform sensitivity analyses on the results.

These issues are illustrated through case studies conducted in the drylands of Latin America.

Selection of criteria and indicators

In the last two decades, criteria and indicators (C&I) sets have found widespread application in the forest sector, and particularly in the framework for Sustainable Forest Management (Stork *et al*., 1997; Mendoza and Prabhu, 2003; Wijewardana, 2008). C&I have been developed under a series of international initiatives, including ITTO, the Pan-European (or 'Helsinki') Process, the Montreal Process, and the Tarapoto, Lepaterique, Near East, Dry Zone Asia and Dry Zone Africa processes (Newton, 2007). However, there are only few examples of criteria sets specifically designed for the identification of forest restoration priorities. There have been some attempts at defining prioritization criteria at global and regional scales (WCMC, 2000; Newton and Kapos, 2003), whereas at a more local level, studies coupling decision analysis and GIS have proposed limited sets of case-specific criteria (Cipollini *et al*., 2005; Marjokorpi and Otsamo, 2006).

A ready-to-use list of criteria that restoration practitioners can refer to and apply in practice is lacking. There is therefore a need for C&I appropriate for prioritizing forest restoration actions at landscape scales, which are readily applicable to different contexts. In order to be useful for the identification of priority sites, C&I should be able to capture spatial variability, given that forest management plans are spatially explicit and today are typically developed and implemented using a GIS (Kangas *et al*., 2000).

Our research on the selection of restoration priorities simultaneously considered areas where restoration is needed (e.g. owing to the presence of endemic species or threats), and areas where restoration is likely to succeed (e.g. owing to soil conditions). This suggested that C&I should belong to two main groups: those that refer to the need for biodiversity restoration (B), and those that refer to the feasibility of the restoration interventions (F) (Orsi and Geneletti, 2010). The first group of C&I is then expected to define where restoration is more urgent for the conservation of biodiversity. The restortation of such areas is expected to preserve habitats (e.g. sites of high biodiversity) and the ecological structures that help maintain connectivity within the landscape (e.g. biological corridors). Conversely, the second group is intended to provide information about the 'restorability' of land (Miller and Hobbs, 2007), which is the ecological cost of successfully achieving the restoration goals. This rationale has been adopted to develop a list of C&I to be used in the identification of forest restoration priorities. The term 'criterion' (C) is used to refer to the general concept (e.g. fragmentation of native forest), while the term 'indicator' (I) is used to refer to an operational way to express or measure a criterion (e.g. edge density, patch density). Both definitions are consistent with SFM C&I processes, such as the Montreal Process (1995).

Identifying C&I through a Delphi survey

The value of expert knowledge for natural resource management is widely recognized: it allows decision makers to take decisions when knowledge based on objective observations is not available (Kangas and Leskinen, 2005; Geneletti, 2008). The Delphi survey is a technique to elicit expert opinion that has been extensively applied to conservation and natural resource management (MacMillan and Marshall, 2006). Delphi surveys aim at soliciting the advice of a panel of experts, and whenever possible forging a consensus. The approach is based on structured and written questionnaires which panellists are asked to answer anonymously. All responses are summarized and reported back to panellists who have the opportunity to revise their judgements.

In order to overcome the lack of an agreed-upon list of C&I to prioritize forest restoration areas, we present here the results of a Delphi survey (fully described in Orsi *et al.,* 2010) conducted with a large group of forest experts that focused on one specific restoration objective: biodiversity conservation. Some 120 experts were identified (of which 37 completed the survey) based on three criteria: personal knowledge, literature review and revision of project databases. In the first round participants were asked to draw preliminary lists of C&I. The results were processed by clustering similar criteria. Criteria defined by similar wording or synonyms were aggregated. Subsequently, a further aggregation was carried out by bringing together criteria that, although defined by non-synonymic words, had the same meaning according to the comments provided by experts. Criteria or indicators not complying with the definitions provided were disregarded. This led to the identification of 20 criteria for factor B, and 10 for factor F (see Annex I). The number of indicators associated with each criterion varied from only one to 18.

In the second round, participants were presented with the revised criteria and indicator list, and were asked to select up to eight criteria from each of the B and F factors and up to three indicators for each criterion. The results were processed by eliminating redundancies and selecting only the most agreed-upon criteria through cut-off thresholds. **Tables 9.1** and **9.2** show the selected criteria and indicators, together with the intra-criterion citation rate, namely the percentage of respondents who selected that specific indicator among the respondents selecting the related criterion. On average, each criterion was linked to 11 indicators. 'Connectivity-corridors' and 'degradation (B)' presented the lowest number of indicators (6), while 'diversity (species level)' presented the highest (17). The citation rate is highly variable within each criterion. Only four indicators were selected by at least 70% of experts: land-use change, linkages between habitat unit, landscape structural diversity, and amount of remnant vegetation.

Table 9.1 Indicators for the B factor as selected by experts. The citation rate is the percentage of respondents who selected a given indicator out of those who selected the related criterion. From Orsi *et al*., 2010.

Criteria	Indicators	Citation (%)
Connectivity-corridors	Linkages between habitat units	70.59
	Presence or absence of wild areas connected to the restoration area	52.94
	Amount of interior habitat within a unit	47.06
	Distance from protected sites	29.41
	Corridor length	23.53
	Corridor width	23.53
Degradation	Land-use change	89.47
	Deforestation rate	47.37
	Fire frequency	36.84
	Soil erosion	36.84
	Road density	21.05
	Pollution indices	5.26
Disturbance	Disturbance classification	65.22
	N. of people depending upon the ecosystem	47.83
	Area of vegetation type after disturbance/area of vegetation type before disturbance	43.48
	Amount of area logged	21.74
	% of invasive species	21.74
	N. of people living within the ecosystem	13.04
	Natural Disturbance Type (NDT) classification	13.04
	% of agricultural area	13.04
	% of populated area	13.04
	Area/perimeter	8.70
	Distance from roads	8.70
	Road density	8.70
	% of area logged by slope class	4.35
Diversity (ecosystem/ landscape level)	Landscape structural diversity	70.00
	Landscape functional diversity	60.00
	Canopy cover	40.00
	Presence or absence of diverse ecosystem at the landscape scale	30.00
	Diversity of soils	20.00
	Presence or absence of water	20.00
	Altitudinal variation	15.00
	Amount of deciduous trees	10.00
	Amount of dead wood	5.00
	Azimuthal variation	5.00
	Quality of dead wood	5.00
Diversity (species level)	N. of endemic species	57.89
	Beta diversity	52.63
	N. of keystone species lost	47.37
	Species richness	47.37
	N. of keystone species	42.11
	N. of major vegetation types	26.32
	Abundance	10.53
	Age	10.53
	Forest density	10.53
	N. of native species / N. of exotic species	10.53
	Evenness	5.26
	Fisher's Alpha	5.26
	N. of birds	5.26
	N. of interactions among species	5.26
	N. of TER species	5.26
	Shannon diversity	5.26
	% live / dead	5.26

Table 9.2 Indicators for the F factor as selected by experts. The citation rate is the percentage of respondents who selected a given indicator out of those who selected the related criterion. From Orsi *et al.*, 2010.

Criteria	Indicators	Citation (%)
Degradation	Amount of remnant vegetation	76.47
	Erosion of topsoil	47.06
	Amount of old-growth trees	41.18
	Compaction	35.29
	N. of remnant tree species	35.29
	Species richness	29.41
	Amount of seed dispersers	17.65
	N. of pioneer species	17.65
	Soil fertility	17.65
	Nutrient depletion	11.76
Disturbance	Land use	59.09
	Fire frequency	45.45
	Amount of herbivory	40.91
	People/km^2	36.36
	Livestock data	22.73
	Presence or absence of invasive species	22.73
	Regeneration ability of invasive species	22.73
	Road density	22.73
	N. of invasive species	9.09
	Presence or absence of pests and diseases in the region	9.09
	Presence or absence of noxious weeds	4.55
	Type of livestock	4.55
Natural regeneration potential	Survival capacity	45.00
	Distance from natural forest	40.00
	Growth potential	30.00
	Presence or absence of biological corridors	25.00
	Distance to seed sources	20.00
	Presence or absence of minimal biotic structures	20.00
	Seedling density	20.00
	N. of seed trees and shrubs	15.00
	Presence or absence of unique genetic variants	15.00
	Rhizomes and root material	15.00
	Distance from protected areas	10.00
	N. of birds	10.00
	Syndromes classification of the landscape unit	5.00

This study provided comprehensive lists of C&I (Annex I), as well as smaller subsets (**Tables 9.1** and **9.2**) that can be used as a reference in studies aimed at selecting forest restoration priorities for biodiversity conservation. In the subsequent stages of the prioritization study, experts and stakeholders can provide further input by assessing the relative importance, or weight, of each criterion (**Box 9.1**). Practitioners can refer to these lists to select on a case-by-case basis the C&I that appear to be most suitable, in the light of the features of the study area and other key factors, such as scale and

Only with the active participation of local people can forest landscape restoration programmes succeed. Here a local workshop on reforestation practices is being conducted in Yanhuitlán in Upper Mixtec Region, Oaxaca, Mexico. Photo: R.F. del Castillo

La Sepultura Biosphere Reserve, Chiapas, Mexico. Photo: N. Tejedor

specific purpose of the restoration intervention and data availability. Additionally, socio-economic variables, such as cost of the intervention and willingness of local population, are likely to be included in the analysis, in order to complement the ecological criteria presented here. The following section presents an application that shows how stakeholders can be involved to select a comprehensive set of criteria for identification of forest restoration areas.

Box 9.1 Weight assessment of forest restoration criteria through interviews with experts in the Upper Mixtec region, Mexico

D. Uribe-Villavicencio, D. Geneletti, F. Orsi, R. F. del Castillo

Expert opinion can provide valuable information about how to evaluate and weigh criteria for the prioritization of restoration sites. A study in the Upper Mixtec region in Oaxaca (Mexico) shows a possible approach to this. The area encompasses three biogeographic zones: the Balsas basin with tropical dry forest, the Sierra Madre del Sur with pine-oak forest and the Tehuacán with xerophyte scrubs. Humans have been living here for about 10,000 years and several small and scattered human settlements are found across the region. Most communities are indigenous and highly marginalized. Agriculture and pasture constitute both the main economic activity and a significant threat to forests.

Ten prioritization criteria were selected based on: easy computability, significant variability over the study area and spatial character. These were both ecological (distance from rivers, risk of soil erosion, distance from forest, slope, insolation) and socioeconomic (population density, distance from human settlements, marginalization index, distance from roads, distance from agricultural fields). Each of the above criteria is likely to have a different relative importance and its possible scores might be interpreted in many different ways with respect to the prioritization goal.

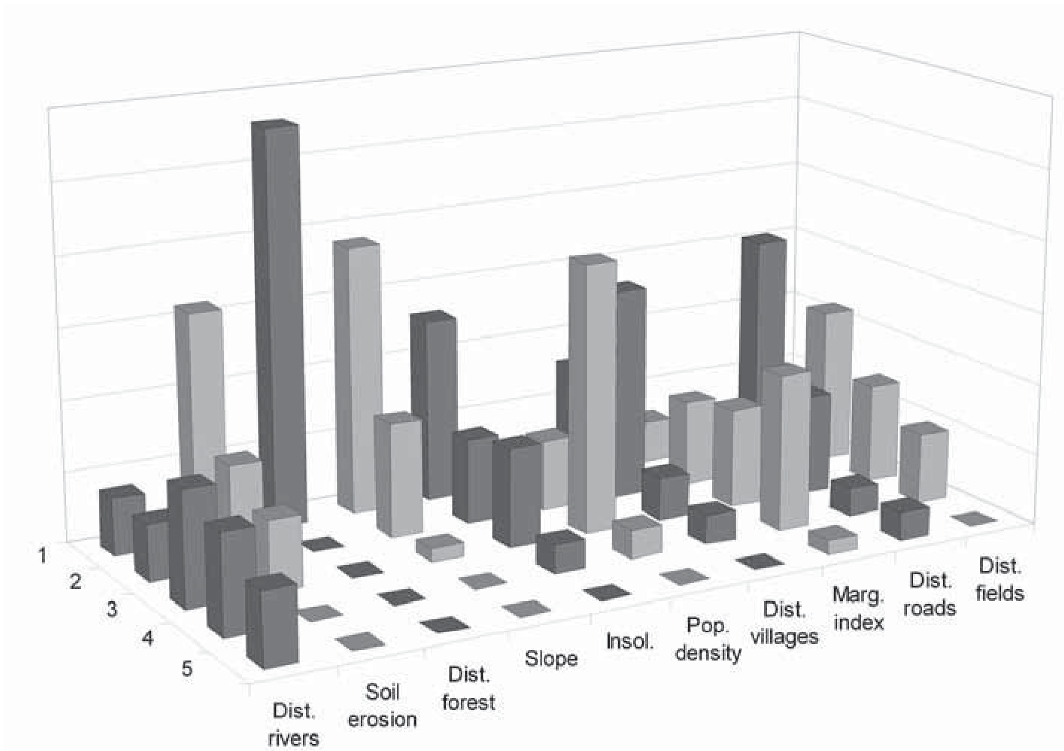
In order to address these issues, 28 people with varied expertise in forest ecology were interviewed. For each criterion, they were asked to provide a weight representing its relative importance (1 = very important, 2 = important, 3 = medium, 4 = little importance, 5 = not relevant), and a cursory evaluation (1 = the lower the score the higher the priority, 2 = the more intermediate the score the higher the priority, 3 = the higher the score the higher the priority). **Fig. 1** summarizes the results of interviews for what concerns the weights. 'Distance from forest' and 'slope' were identified as the most important criteria. Experts assigned higher restoration priority (i.e. higher desirability) to areas exposed to sun, close to forests and rivers, far from human settlements, roads and agricultural fields, with moderate slope and low population density.

A map for each of the 10 above-listed criteria was generated using basic GIS techniques. Their values were then converted to a common range 0–1 according to the experts' evaluations (0 = lowest desirability; 1 = highest desirability). The obtained maps were summed up through a weighted linear combination by introducing the weights proposed by experts to obtain a final suitability map. In total, 28 maps were generated by varying the evaluation and weights according to the experts' opinions. Priority sites were identified by introducing a suitability threshold and filtering out patches smaller than 5 km^2 (negligible at this scale). A final comprehensive map was obtained as the combination of all 28.

While showing how to effectively involve experts in the evaluation of prioritization criteria, this study has also highlighted the considerable uncertainty that affects such evaluations. To this extent, the involvement of stakeholders and meetings with local communities can suggest whether a restoration plan is as good on the ground as it is on the map.

Box 9.1 (cont.)

Figure 1 Histogram that summarizes the expert opinion about weights. For each criterion (represented by a different colour) five bars show the opinion of the 28 experts according to the five classes of importance (1 = very important; 2 = important, 3 = medium, 4 = little importance, 5 = not relevant). 'Distance from forest' and 'slope' are the most important criteria as shown by high bars in the first class.



Involvement of stakeholders: a study in northern Argentina

Stakeholder analysis is an approach for understanding a system by identifying the key actors, and assessing their respective interest in that system (Grimble and Chan, 1995). Renard (2004) defines stakeholder analysis as a focused and well-planned exercise aimed at answering questions that are directly relevant and useful to the planning and management process. Stakeholder analysis has been applied to several fields of study, including business management, international relations, policy development, participatory research, and increasingly often natural resource management. The incorporation of different stakeholders' values and concerns is a critical factor in forest restoration. In recent decades, the focus of forest management has gradually widened from sustained yield timber management to sustainable forest ecosystem management. This has occurred on the basis of increasing recognition of multiple forest values beyond the customary timber values, such as the value of non-timber products and ecosystem services (Toman and Ashton, 1996). Forest management has also changed from 'management by exclusion' to 'management by inclusion', recognizing the need to incorporate in forest management decision-making the preferences of multiple stakeholder groups, such as local communities, environmental groups, forest industries and aboriginal groups (Buchy and Hoverman, 2000; Sheppard and Meitner, 2005).

The Forest Landscape Restoration (FLR) approach has organized all these principles in a structured framework that also accounts for the main principles of the ecosystem approach (EA): acting at the ecosystem scale, involving stakeholders, considering alternatives (ITTO/IUCN, 2005). We applied the principles of EA and FLR to the identification of forest restoration priorities in the provinces of Salta and Jujuy, northwestern Argentina, an area covered by the Yungas, an extensive system of native forests. Forest protection and restoration is crucial for the Yungas ecosystem: these forests are considered to be an international hotspot for biodiversity. In northern Argentina the conservation of the Yungas is mainly threatened by the expansion of the agricultural frontier. More than 194,000 ha were deforested from 1998 to 2002 (Gasparri *et al.,* 2004). This study (described in detail in Ianni and Geneletti, 2010) represented the first attempt to apply EA principles to forest restoration at landscape scale in the Yungas region. We identified the social actors that had a stake in the forest management and we involved them in multicriteria analysis sessions aimed at identifying the priority area for forest restoration interventions. The combination of all of the alternatives at stake in the analysis represented the Yungas landscape mosaic, as recommended by the FLR approach.

A participatory multicriteria approach

The approach is based on three steps:

1. Identification of the social actors that have a stake in the forest management of the study area;
2. Identification of feasible restoration actions, and criteria to prioritize them; and
3. Comparison and ranking of restoration actions according to social actors' needs and expectations, using multicriteria evaluation (MCE).

Yungas stakeholders were identified as local actors that had a stake in each of the ecosystem services provided by the forest (production, regulation and cultural services). Stakeholders represented different organizations and economic sectors working in the Yungas forests, as shown in **Table 9.3**. Their opinions were collected through semi-structured interviews, and then represented in cognitive maps. A cognitive map links concepts to form chains and aims at disclosing individual perceptions of consequences and explanations associated with concepts (Eden and Ackermann, 2004). Özesmi and Özesmi (2004) defined cognitive maps as qualitative models of how a given system operates. The cognitive maps that were constructed basically provided the answer to the following question: what is, in your view and experience, forest restoration? **Fig. 9.1** shows the cognitive map that represents the view of the stakeholder group 'timber industry', which exploits the forest to get commercially viable timber. According to this group of stakeholders, forests have no intrinsic value: the value of a forest exists as long as commercially valuable timber can be extracted. Forestry entrepreneurs use the term 'enrich' to indicate the plantation of both native and exotic species in native forests. (They always use the word 'exotic'; they never use the word 'invasive').

Table 9.3 Yungas stakeholders were identified as the local actors that had a stake in each of the ecosystem services provided by forest (production, regulation and cultural services). Stakeholders represent different organizations and economic sectors working in the Yungas forests. From Ianni and Geneletti, 2010.

Resource	Services	Related goods and services	Stakeholders
Native Forest	Production	Fodder (including grass from pastures) Fuel (including wood and dung) Medicinal resources Commercially viable timber Genetic resources	Local communities Timber industry Forestry engineers Economy consultants
	Regulation	Gas regulation Climate regulation Disturbance prevention Water regulation Water supply Soil retention Soil formation Nutrient cycling Pollination	Environmental agencies Governmental agencies
	Cultural	Habitat for wild plant and animal species Cultural, historical and religious heritage Scientific and educational information Opportunities for recreation and tourism	Community-based rural tourism operators

Workshop in the community, Oaxaca, Mexico. Photo: R.F. del Castillo

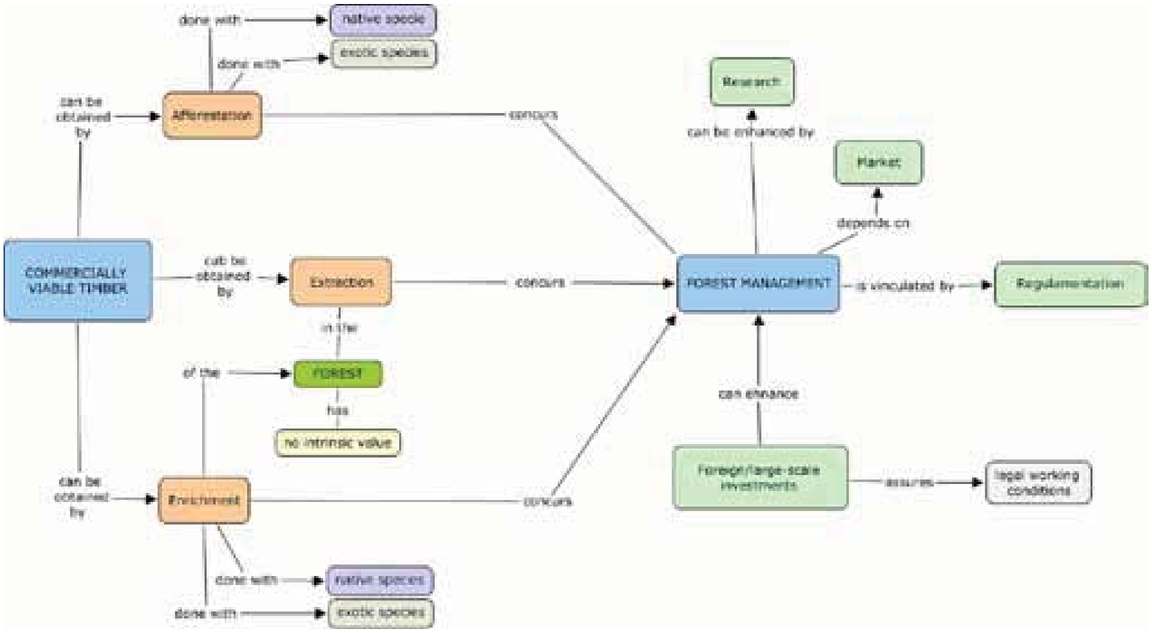


Figure 9.1 Cognitive map of timber industry stakeholder group. According to this group of stakeholders, forests have no intrinsic value: the value of a forest exists as long as commercially viable valuable timber can be extracted. From Ianni and Geneletti, 2010.

Subsequently, a stakeholder workshop was held to identify the alternative restoration options, as well as the criteria to be used in the selection of the most suitable option. Not all of the interviewed stakeholders participated in the workshop, which was attended by two biologists with extensive working experience in the Yungas ecosystem; an economist with expertise on the economic valuation of forest plantations; the coordinator of the regional office of natural resources; a sociologist with expertise in community-based rural tourism; the coordinator of the regional office for indigenous rights; and an engineer consultant for many forestry entrepreneurs of the area. Stakeholders represented five extremely different perspectives, concerning cultural background, interests and vision.

During the workshop, stakeholders were first asked to identify alternative restoration actions, i.e. potential areas for implementing FLR. Ten alternative *fincas* (estates derived from the Spanish land tenure system) were proposed, differing in relation to management, land use, land tenure, and size. The *fincas* belong to indigenous communities (Yaquy, Los Naranjos, Finca Vinalito), an agro-industry (Ledesma), private owners (agricultural lands), the state (Finca Chalican, Finca Acambuco), private forestry companies (Finca Fontanelas, Finca Rio Seco, Finca El Pongo). Twenty criteria were selected and classified into four groups: biophysical, social, economic and political criteria. The assessment of these criteria was qualitative (e.g. ++,--), quantitative (e.g. km, pesos/ha) or binary (yes/no), according to data availability. Quantitative criteria were classified as 'benefit' (B, i.e. the higher the value, the more suitable the option) or 'cost' (C, i.e. the lower the value, the more suitable the option). **Table 9.4** presents the criteria and their unit of measurement. After the identification of the criteria, participants were asked to rank the groups of criteria, as well as the criteria within each group, by using a three-level qualitative scale: each criterion can be (a) equally important, (b) slightly more important or (c) strongly more important than the one that follows in the ranking.

Table 9.4 Twenty criteria were selected and classified into four groups: biophysical, social, economic and institutional. The assessment of these criteria was qualitative (e.g. ++,--), quantitative (e.g. km, pesos/ha) or binary (Yes/No), according to data availability. Quantitative criteria are classified as 'benefit' (B, i.e. the higher the value, the more suitable the option) or 'cost' (C, i.e. the lower the value, the more suitable the option). From Ianni and Geneletti, 2010.

	Criterion	Unit	Description
Bio-physical	Forest recruitment (C)	Number of plants	Number of new plants per hectare (indirect estimation of the good health of the forest)
	Degradation trend	–, ++	Evolution of the forest in absence of any intervention
	Biodiversity	0, +++	Intrinsic value of the site concerning biodiversity
	Ecological added values	–,+++	Ecological features of the landscape that can concur to achieve a successful result
	Soil conservation	0,+++	Additional benefit: capability of afforestation/ enrichment to preserve or recuperate the soil
	Water availability (B)	Mm	Measure of the quantity of rain
Social	Level of organization	0, +++	Measure of the organization and of the flow of information (decision) in the community/company
	Concern for the forest	–,+++	Spiritual value for a community or a concern of a private entrepreneur
	Expertise in forest plantations (B)	Years	Years the community/company has been involved in forestry activities
	Availability of persons working in plantations (B)	Number of people	Number of persons that would actively be involved in the project
	Level of interest/ engagement	0,+++	Level of interest demonstrated by the community/company for the restoration project
	Level of conflict	0,+++	Level of conflict in the community/company that could affect the successful implementation of the project
	Juridical security	0, +++	Land tenure
Economic	Logistic facilities (C)	km	Distance from logistic headquarters
	Operational cost (C)	pesos/ha	Cost of the facilities
	Number of beneficiaries at medium step (B)	Number of people	Number of persons that could benefit from the project (i.e. families of the workers)
	Probability of land-use change (C)	%	Probability the community/private company will change land use in the future
Political	Visibility	0,+++	Visibility of the project in the region
	Potential replicability (B)	yes/no	Possibility for the community/company to replicate the project
	Resilience	–,+++	Capability to cope with unexpected events (instability, unpredictable changes, etc.)

Finally, the ten alternative options were assessed using the information provided by each of the stakeholders and by aggregating the criteria through a weighted linear combination. Sensitivity analysis was then performed to assess the stability of results, their dependence on the set of weights selected by the stakeholders and the similarity between alternatives. **Fig. 9.2** shows the ranking of the alternatives according to the view of the five stakeholders. As it can be seen, *finca* Ledesma, Los Naranjos and Rio Seco have the highest performance. The results showed robust ranking and no rank reversal in almost all of the cases. The sensitivity analysis showed that although different groups of stakeholders diverged on the weights assigned to the criteria, they all agreed upon the dominance of the Ledesma and Los Naranjos *fincas* over the others. In particular, the Ledesma *finca* appeared to be the most suitable option, by virtue of its good environmental performance and very good social and political performances. Ledesma spans over wide areas, it borders Calilegua National Park and the Yungas Biosphere Reserve; the soil is fertile and there is relatively high water availability. A successful restoration activity is therefore expected to succeed. In the social set of criteria the criterion that describes the level of organization indicated high performance of the Ledesma agroindustry. In fact, in an industry, decision making is typically hierarchical and follows a command and control approach. This minimizes conflicts and maximizes the probability of the successful implementation of a restoration project over time. From a political point of view, *finca* Ledesma ranks first because it is the largest factory of the region; a restoration activity there would have great visibility in the region.

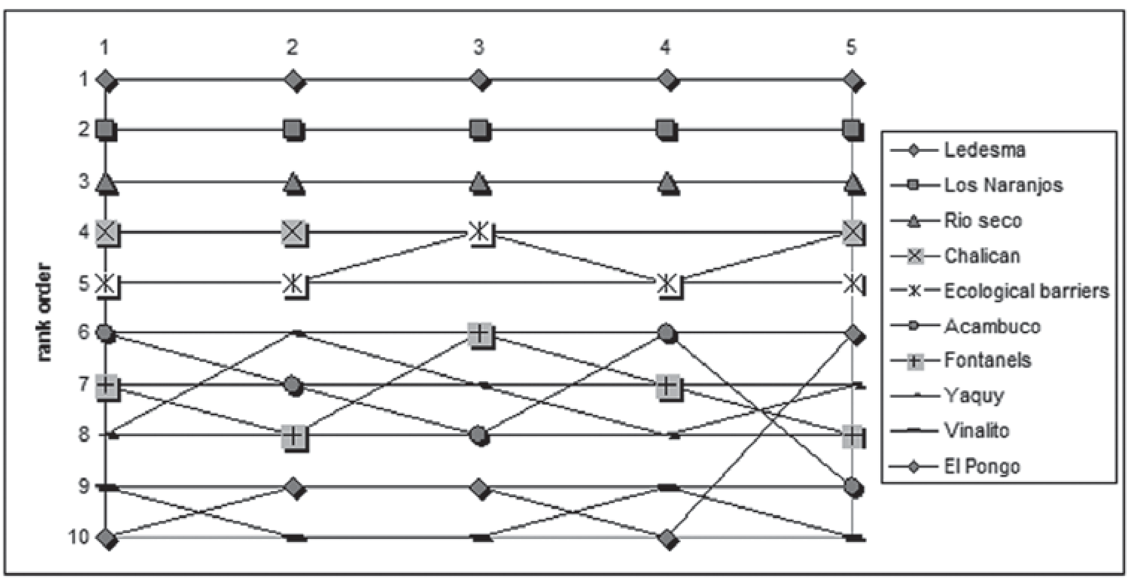


Figure 9.2 Ranking of the ten forest restoration options according to the perspectives of the five stakeholder groups:

1: Aboriginal community
2: Forestry consultants
3: Environmental NGO
4: Timber industry
5: Governmental agency
From Ianni and Geneletti, 2010.

Lessons learned

The outcome of the implementation of restoration strategies for degraded primary forests basically depends on three issues (Sayer *et al.*, 2005): a technical factor, i.e. the condition of the forest stand, and two wider social factors, i.e. the objectives of the restoration programme and the actors involved. This case study stressed the importance of clearly setting the objective of the restoration programme and demonstrated the effectiveness of MCE methods in involving stakeholders. Here are some lessons we learned from the activity we carried out in northern Argentina:

- The stakeholder analysis was useful to understand stakes and attitudes of potential users of FLR. When deciding to implement a restoration project, it is not trivial to pose the question: what is forest restoration? What are its purposes? This is very relevant to the intervention because, as stated in the key principles of FLR, an integrated forest management involves a package of solutions that can be proposed.
- MCE helped to combine different sources of information and to structure a transparent evaluation approach (see also **Box 9.2**). It also served as a forum for discussion, negotiation, exchange of knowledge and final selection of a *finca* that was closest to the economic, social, environmental and institutional criteria, as perceived by the relevant actors involved in the decision. The public MCE process seemed to work well and was favourably supported by various stakeholder comments. The explicit structure of the decision problem was particularly appreciated. The structured process, and the avoidance of seeking consensus as the end-goal, seemed consistent with stakeholders' expectations.
- MCE is run by people. In theory, the more different background and social positions participants have, the more successful is the approach. The core of MCE lies in the participation of different actors, hence we should be fully aware that people's values and beliefs heavily condition the analysis. In our case, scepticism about real impact of the method on concrete decisions and actions remained evident among the stakeholders. The MCE logic was very unfamiliar to some of the participants and misunderstandings occurred, also because of time constraints. We recommend that sufficient time be allocated for MCE so that people with different skills and expertise can familiarize themselves with it, and understand its contribution to decision making: it is a tool to improve the process, rather than provide the solution.

Box 9.2 Use of biotic, abiotic and cultural variables for tropical dry forest conservation and restoration in central Veracruz, Mexico

C. Gómez Alanis, G. Williams-Linera

In central Veracruz, there are little-known sites with remarkable features that may contribute enormously to the knowledge and conservation of the biological and cultural-historical patrimony of Mexico. Central Veracruz has been heavily influenced by human activities (mainly agriculture and cattle production), to the extent that only 9.26% of original forest remains, and it is highly fragmented. The vegetation that remains is a biodiversity refuge and includes an important corridor for one of the largest annual migrations of raptors in the world (the 'River of Raptors'). Moreover, this area has numerous remains of pre-Hispanic

Box 9.2 (cont.)

settlements (600 to 1500 A.D.) and played an important role in the Mexican independence period (19^{th} century).

The study area covers 1084 km^2, mainly in the municipality of Paso de Ovejas and parts of the adjacent municipalities of Comapa and Puente Nacional. This study was designed to identify priority areas for the conservation of TDF fragments and cultural elements from a landscape perspective. The objectives were to determine the spatial relationship of forest fragments and sites with cultural-historical significance as well as to propose ways to connect priority forest areas to cultural-historical landmarks, via both protection and restoration of ecological corridors.

We used three types of indicator or group of simplified variables for a multicriteria analysis:

(1) Biotic indicators are plants and animals found in forest fragments and considered endemic or threatened species. These species fell into the category of national or international protection or conservation status. The list of tree, mammal, amphibian, reptile, and bird species was obtained from electronic databases and was enriched by a survey of current literature and field verification. In addition, key informants were interviewed about the regional biota using pictures of the species recorded in the area. The species listed included 82 tree species (4 endemic, 2 protected), 64 mammal species (5 protected), 29 reptile species (4 endemic, 5 protected), 25 amphibian species (8 endemic, 8 protected), and 111 bird species (2 endemic, 27 protected).

(2) Abiotic indicators are the physical characteristics of forest fragments such as core area, fragment size, isolation, distance to human infrastructure, influence of human populations and mean population size.

(3) Cultural-historical indicators were architectural structures (55) and isolated mounds (600 to 1500 A.D.), paintings on cave walls and ceilings (17), archaeological carved stones (3), constructions belonging to the Royal Road (2 bridges, 1 fort), and 3 estates or ex-haciendas from the 19^{th} century.

The analysis and evaluation of the criteria led to the identification of the priority areas for conservation and restoration. These have a unique combination of characteristics, such the highest core area, increased distance from infrastructure and towns, lowest population density in the area, and number of cultural-historic elements within a buffer zone of 1000 m from the edge of each fragment. Also, they had to be less than 13 km apart and located within a landscape matrix of secondary vegetation, other forest fragments, tree plantations, and landscape elements such as permanent or intermittent streams that facilitate connectivity between selected areas.

Mapping of these criteria illustrated a concentration of abiotic, biotic, and cultural indicators in five areas (C1-C2, A1-A2, X1-X2, H1-H2—MM, and PN, see **Fig. 1**). C1-C2, A1-A2, and X1-X2 are important because of their high biodiversity, greater distance to main infrastructure and towns, diversity of cultural-historical elements, and matrix appropriate for connectivity maintenance. H1-H2-MM showed a greater and less disturbed forest core area and many archaeological sites. PN deserves special mention owing to its cultural diversity and endemic or protected species; however, the high level of anthropogenic disturbance decreases viability for conservation and restoration of biotic resources but not for conservation of cultural-historical ones. Ecological restoration should be used to connect cultural sites and create routes for ecotourism that may provide an economically sustainable alternative activity that could ensure the well-being of local populations.

Box 9.2 (cont.)

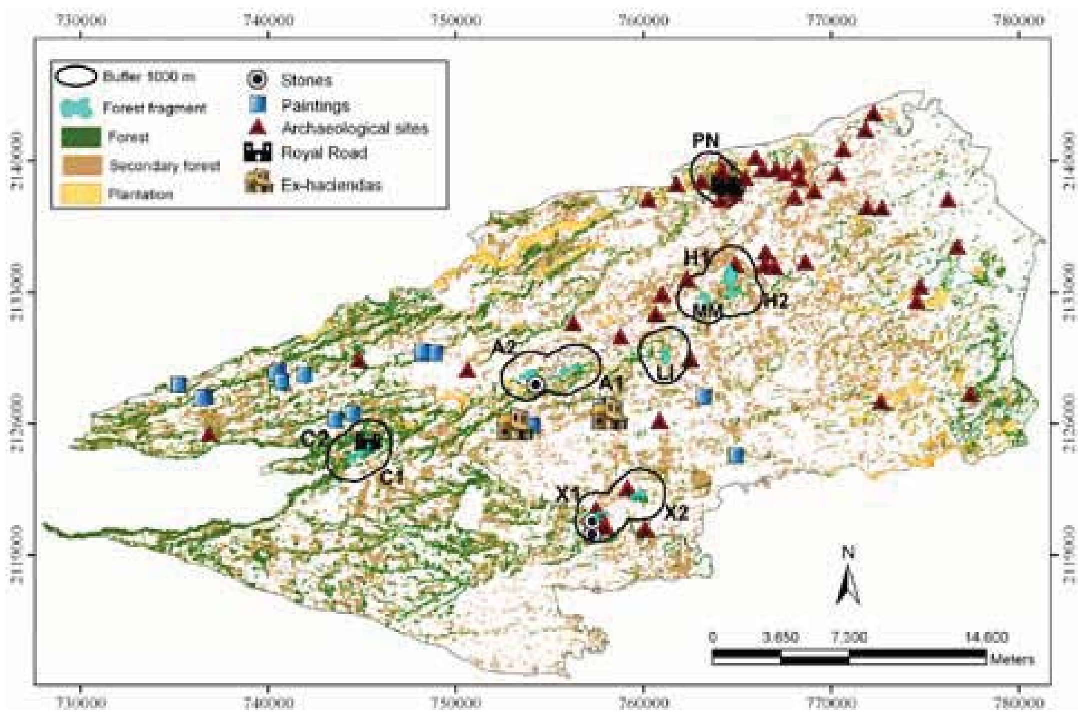


Figure 1 Map showing the different layers of information used and the buffer of 1000 m from the edge of tropical dry forest fragments in central Veracruz, Mexico. Forest fragments are: PN, Puente Nacional; H1, Hato los Marines 1; H2, Hato los Marines 2; MM, Mata Mateo; LI, El Limón; A1, A2, Acazónica; X1, X2, Xocotitla, C1, C2, Dos Caminos.

Use of spatial decision-support tools: a study in Chiapas, Mexico

Whether a site should be accorded priority for restoration clearly depends on the objectives of the restoration process. For example, a restoration plan aimed at enhancing landscape connectivity would identify different priorities than a plan aimed at increasing the provision of timber to human settlements. Restoration priority can be seen as a function of two factors (see earlier): B, which represents the need for biodiversity restoration (where should forest be restored?), and F, which represents the feasibility of restoration (where is restoration likely to succeed?). As the focus initially is on the ecological issues, B refers to the identification of those areas that play a major role in the conservation of biodiversity (e.g. species richness, core habitats), while F considers the ecological obstacles to a successful restoration. A suitability map for each of the B and F factors has to be generated that represents the restoration priority of land according to its need for and feasibility of restoration, on the basis of a set of criteria that can be spatially represented. If raster maps are used, the basic units of analysis are the individual cells, and this gives the user enough freedom to shape effective restoration sites. However, other basic units of analysis (e.g. municipalities, watersheds) can also be used (see **Box 9.3**), as well as multi-step approaches in which the level of detail of the basic units increase at each step (see **Box 9.4**). The selection of most suitable areas from the suitability maps can be performed by means of threshold values.

Box 9.3 Selecting forest restoration priorities at the watershed level in central Chile

R. Fuentes, A. Miranda, C. Echeverría, C. Smith, I. Schiappacasse

Using watershed as the basic unit of analysis to identify forest restoration priorities is a good choice when working at large scales. We conducted a study in an extensive area of central Chile: 13,000 km^2, including 59 municipalities in Valparaiso, Libertador Bernardo O'Higgins and Metropolitana Administrative Regions, and hosting around 30% of the national population. The native vegetation is dominated by *espinales*, a savannah-like vegetation of *Acacia caven* and by sclerophyllous endemic species. The main drivers of deforestation and fragmentation are the extraction of firewood, agriculture and the introduction of exotic herbivores and cattle.

The study area was divided into 30 watersheds, whose prioritization was based on environmental and socioeconomic criteria. The set of criteria and their relative importance (weight) were identified during workshops with stakeholders, interviews with local governmental and non-governmental organizations, meetings of project researchers and interviews with representatives of local communities (**Fig. 1**).

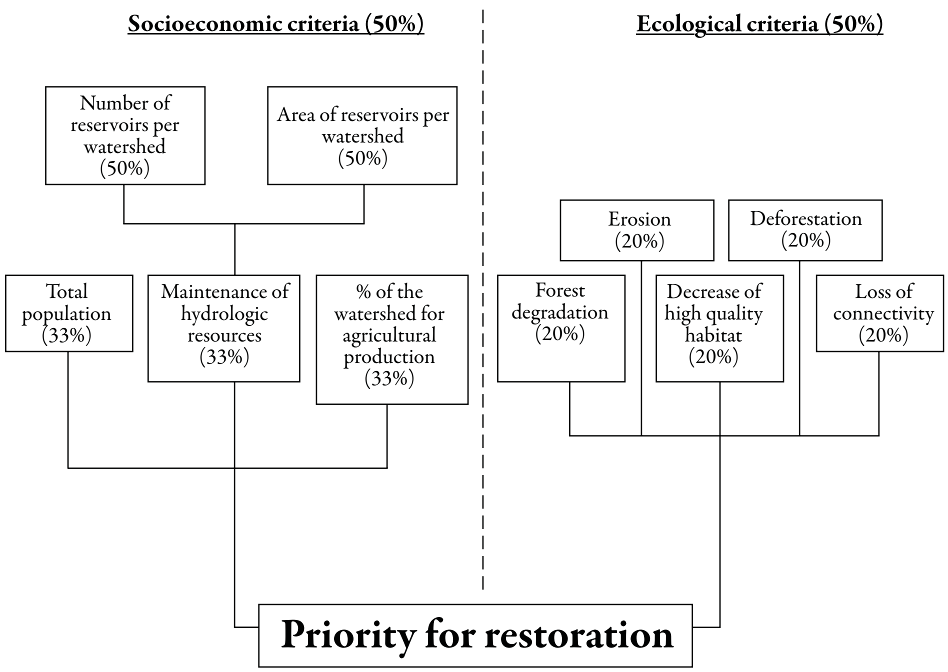
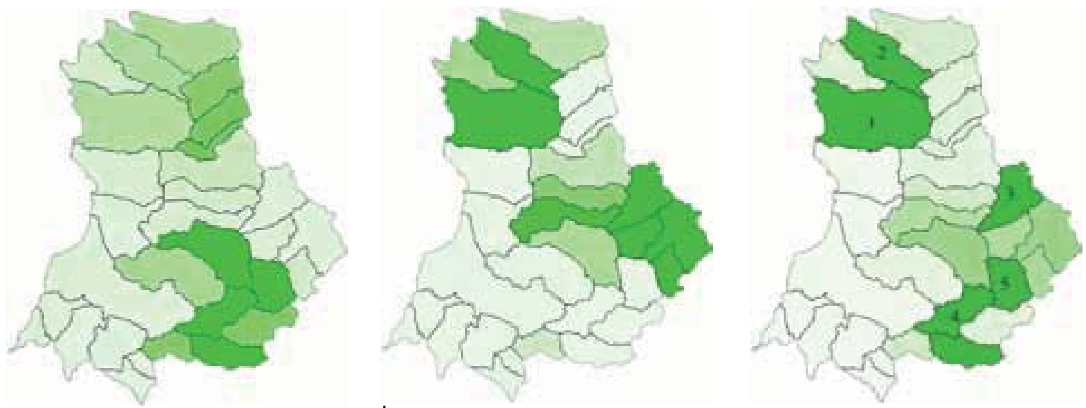


Figure 1 The criteria tree developed during workshops with stakeholders and interviews with governmental and non-governmental organizations and local communities. Percentages in brackets represent the relative importance (weight) of each criterion.

The criterion maps were generated in a GIS environment, and then combined through weighted summation, by using socioeconomic criteria, environmental criteria and both types of criteria. As can be seen, watersheds with high priority in terms of socioeconomic criteria tend to have low priority when considering environmental criteria, and *vice versa* (**Fig. 2**). This emphasizes the high degree of human intervention that the study area has undergone: the watersheds that sustain the most human-related activities have been heavily degraded well before 1975. The watershed with the overall highest priority (located in Casablanca municipality) is characterized by a vast proportion of agricultural land and a high number of dams, and it has experienced considerable deforestation, resulting in loss of habitat and soil erosion.

Box 9.3 (cont.)

Figure 2 Maps showing the prioritization of watersheds using socioeconomic criteria (a), environmental criteria (b), and both types of criteria (c). Dark green indicates higher priority, light green indicates lower priority. Numbers in map (c) indicate the ranking of the five most important watersheds for restoration. Priorities change when different sets of criteria are considered (maps a and b).



a b c

This case study showed that the prioritization of watersheds is efficient in quickly providing decision makers with information that can be easily understood and can represent a valid starting point for setting policy at regional level. However, while a watershed-level analysis informs the user about which watershed globally deserves more urgent restoration, it does not provide information on the specific conditions within the watershed. This requires on-site analysis of the selected watersheds, so as to identify the most suitable restoration interventions, as well as their location.

Box 9.4 Priority areas for implementing the CDM to forest restoration projects in conservation corridors of the Andes

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Afforestation and reforestation projects are two of the measures included in the Kyoto Protocol within the Clean Development Mechanism (CDM) framework. These measures appear to be a very cost-effective strategy for mitigating climate change. Furthermore, they have great potential in neotropical countries owing to the large amount of land available and suitable for reforestation, and the many benefits they can bring in social and environmental terms. Significant opportunities have been created with the launch of the international market for emission reduction of greenhouse gases (GEI), which led to this study. The study aims were to identify and develop criteria for selecting priority areas for the implementation of CDM, which could also contribute to the restoration of biological corridors in the Andean hotspots (**Fig. 1**).

Potential areas were selected by using specific regulatory, eco-physiological and socio-economic evaluation criteria. In addition, the technical and economic viability of CDM projects was evaluated, for those that are going to contribute to biodiversity conservation in the chosen areas. The proposed work is the first step of a top-down approach, where strategic areas are identified based on coarse-scale information to reach potential sites for pre-feasibility studies with more detailed information. The indices of potential priority areas for the CDM projects identified four locations with potential to evaluate the pre-feasibility. For the Norandean hotspot the chosen places were located in the north zone of the Department of Cundinamarca (Colombia). This zone includes important areas for

Box 9.4 (cont.)

conservation such as The Natural National Park of Chingaza, The Natural National Park of Sumapáz and El Páramo de Guerrero. For the Cóndor Kutukú the chosen places were located in the north zone of San Maritín Province (Peru), which includes the Awajun and Nueva Cajamarca Regions, and the central zone, which is located in La Paz Province and to the west of El Beni Province in Bolivia. The regions of Sabanas de Apolo and Caranavi were also taken into consideration in the study area (**Fig. 2**).

Figure 1 Geographical location of hotspots in study: (1) Choco-Manabi, (2) Norandino (3) Guiana, (4) Tumbes, (5) Condor-Kutukú, and (6) Amboró Vilcabamba (Source: Conservation International) (Source vegetative cover: Eva *et al*. (2004)).

Within the selected areas a feasibility exercise was carried out, which enabled determination of the potential of eligible areas within each region. A cost-benefit analysis provided different market scenarios relating to the price of a tonne of carbon. The forestry model for determining the carbon sequestration potential of the regions was based on reports of the Intergovernmental Panel on Climate Change (IPCC). It was generally concluded that most market scenarios evaluated for the establishment of a forestry project under the CDM have a high possibility of being viable ecologically, economically and socially.

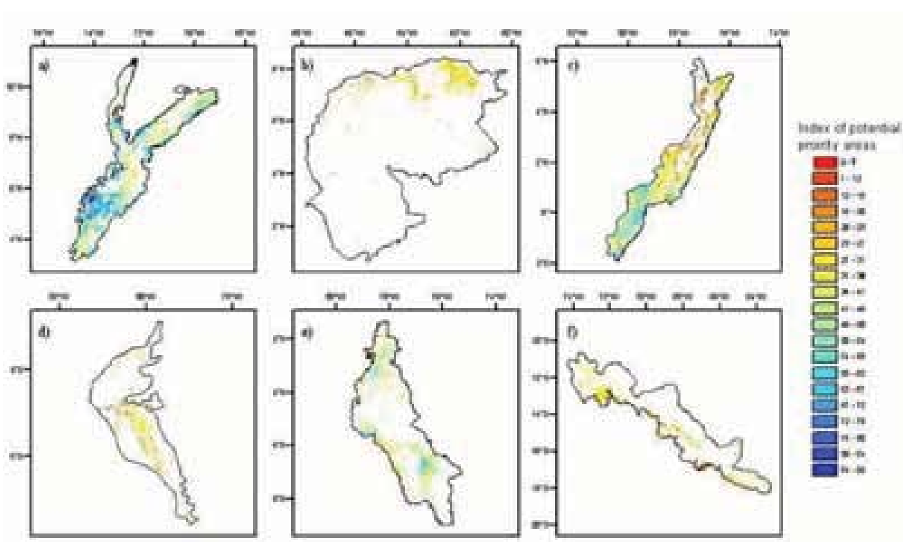

Figure 2 Distribution of potential priority areas (IAPP) for CDM forestry projects in hotspots: (a) Norandino, (b) Guyana Shield, (c) Choco-Manabi, (d) Tumbes, (e) Condor-Kutukú,(f) Amboró Vilcabamba.

This approach, which is thoroughly described in Orsi and Geneletti (2010), was applied to an area of about 18,500 km^2 in central Chiapas (Mexico) to identify reforestation priorities (other forms of restoration were not studied owing to a lack of baseline data). The region, whose elevation ranges from 0 to 2500 m, hosts three main forest types: tropical dry forest (Selva Baja Caducifolia), pine-oak and cloud forest. Two Biosphere Reserves are found in the Region: El Triunfo and La Sepultura. Humans live in several small villages, scattered across the whole area. Agriculture and cattle ranching constitute the main economic activities and are potentially a threat to forest conservation as agricultural fields and pastures are expanded by converting forests.

The following criteria were selected to support the evaluation of the B factor:

- Distance from ecological corridors: ecological corridors, allowing species to move over the landscape, are one of the key components of nature conservation plans and their reforestation may help reducing species isolation;
- Distance from existing forest: areas around existing forests are a priority for their proximity to reservoirs of native species;
- Distance from protected areas: protected areas are a sample of a region's biodiversity to which they provide protection from external threats. Reforesting in and around a protected site means both enhancing the forested ecosystem and creating a buffer zone that prevents the site from being disturbed;
- Tree species richness: sites characterized by high numbers of species are the main target of a reforestation process aimed at conserving biodiversity.

The following criteria supported the evaluation of the F factor:

- Distance from agricultural fields: areas around existing agricultural fields are more likely to undergo land-use change;
- Distance from roads: roads are a source of disturbance as they allow people to have access to nearby areas;
- Distance from urban areas: towns and villages represent a high concentration of human activities, which demands resources from the surroundings;
- Risk of soil erosion: soil degradation can undermine the success of a restoration intervention.

The application of spatial MCE

A raster map was generated for each of the above criteria by means of basic GIS operations (e.g. distance calculation). All criterion maps were combined in a multicriteria fashion. In order to make them comparable, a value function was assessed for each criterion. Value functions transform the score of a given criterion into values between 0 and 1, where 0 corresponds to minimum desirability and 1 to maximum desirability. They also show whether a criterion is a cost (the higher the score the lower the desirability) or a benefit (the higher the score the higher the desirability), as shown in **Fig. 9.3**. Value functions were assessed such that the most suitable areas to locate reforestation interventions were biologically diverse areas within or around ecological corridors, forests, and nature reserves; not in close proximity to agriculture, roads and settlements; and in areas not exposed to severe soil erosion. Once maps had been standardized through value functions, two suitability maps, one representing the B factor and one representing the F factor, were generated by simply adding the maps of the two groups.

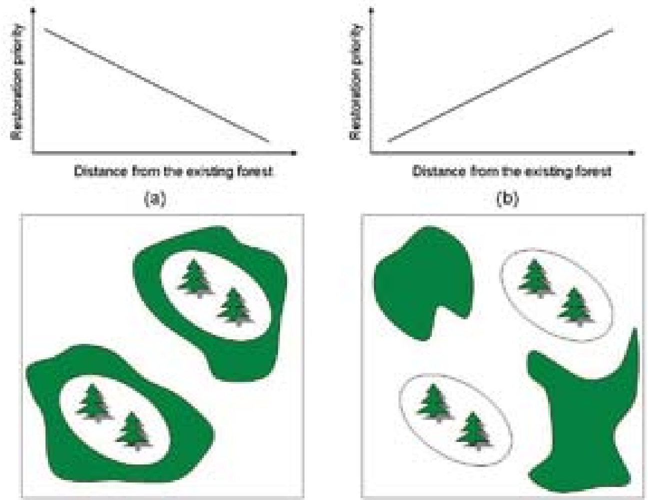

Figure 9.3 Assessment of value functions for the criterion 'distance from the existing forest'. The value function is the mathematical representation of human judgement over a given criterion, informing about the desirability of each score of the same criterion. In this example, function (a) accords higher restoration priority to areas in close proximity to existing forest, whereas function (b) accords higher restoration priority to areas far from the existing forest. The assessment of value functions is a crucial part of decision making and the related uncertainty can result in significantly different decisions.

The selection of most suitable areas from the suitability maps is commonly done by means of thresholds. When two suitability maps are simultaneously considered, thresholds allow both the most suitable cells to be extracted and a non-compensatory approach to be applied. The latter, referring to this specific case, means that a site should be accorded priority for reforestation if it is simultaneously in strong need of reforestation and likely to make the reforestation succeed. Thus, the two suitability maps (B and F) were crossed to obtain information about the number of cells corresponding to any pair of values in the suitability maps. This information would then allow the computation of cumulative information: that is, for any pair of thresholds (one for B and one for F), the number of cells with values for B and F above both thresholds. After having set the reforestation demand (i.e. the total area that the decision maker is willing to reforest) and the minimum threshold values for B and F, it was possible to extract several reforestation options that all achieve the expected demand and fulfil the minimum suitability thresholds. The number of options was eventually reduced by filtering out small patches of contiguous cells. The definition of a total reforestation demand of 15,000–17,000 ha, and the selection of 0.6 as the minimum suitability threshold for both maps B and F resulted in 14 reforestation options (**Fig. 9.4**). The 14 options combined covered an area of about 28,000 ha, of which one-fourth was common to all options. About 2,500 ha were selected by only one option and about 11,000 ha by no more than five options.

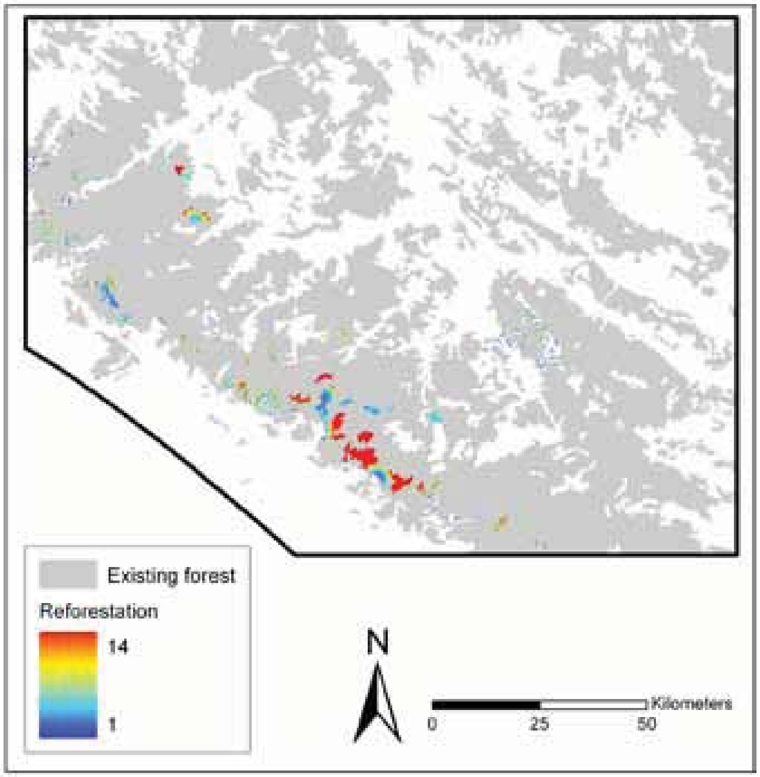

Figure 9.4 The location of reforestation areas according to the 14 reforestation options generated through the proposed methodology. Red colours indicate areas selected by most options, whereas blue colours indicate areas selected by few options only. The total area selected for reforestation is around 28,000 ha of which one-fourth is selected by all options. Adapted from Orsi and Genel-etti 2010.

The options obtained were then compared by introducing additional criteria. These included socioeconomic criteria to select the option that was believed to bring in the maximum benefits to both nature and people. Moreover, these criteria accounted for the spatial configuration of options and therefore they could be assessed only once such options had already been designed. The following criteria, along with the indicators used to measure them, were selected:

- Ecological criteria:
  - Fragmentation of the landscape (Edge Density).
  - Average compactness of forest patches (Mean Shape Index).
  - Enhancement of ecological corridors (reforestation area occurring within ecological corridors).
- Socioeconomic criteria:
  - Land-use conversion cost (reforestation area occurring within agricultural fields).
  - Reduction of soil erosion (reforestation area occurring in soil with intermediate erosion risk).
  - Improvement of livelihoods (reforestation area occurring in poorest regions).

The rationale for these criteria is that the most suitable reforestation option should minimize landscape fragmentation (criteria I and II), improve ecological networks (criterion III), minimize conflicts with agricultural land uses (criterion IV), contribute to reduce soil erosion (criterion V), and improve local livelihoods (criterion VI). The assumption for the use of the criterion VI is that reforesting poorer regions is likely to improve ecosystem conditions and the provision of ecosystem services exactly where people are more vulnerable to ecosystem degradation. Scores showing the performance of different reforestation options against each criterion were computed and subsequently combined in a multicriteria way. Again, value functions were applied to convert criterion scores to a common value range (0–1). Weights, showing the relative importance of each criterion, were assigned to both the single criteria and the criteria groups (ecological and socioeconomic), according to three evaluation perspectives (balanced, environmentoriented, socioeconomic oriented), as presented in **Table 9.5**. The results of the three evaluations are shown in **Fig. 9.5**. Option 10 was considered the most preferable one, in the light of its performance under the three evaluation perspectives, and its suitability according to both ecological and socioeconomic variables (**Fig. 9.6**).

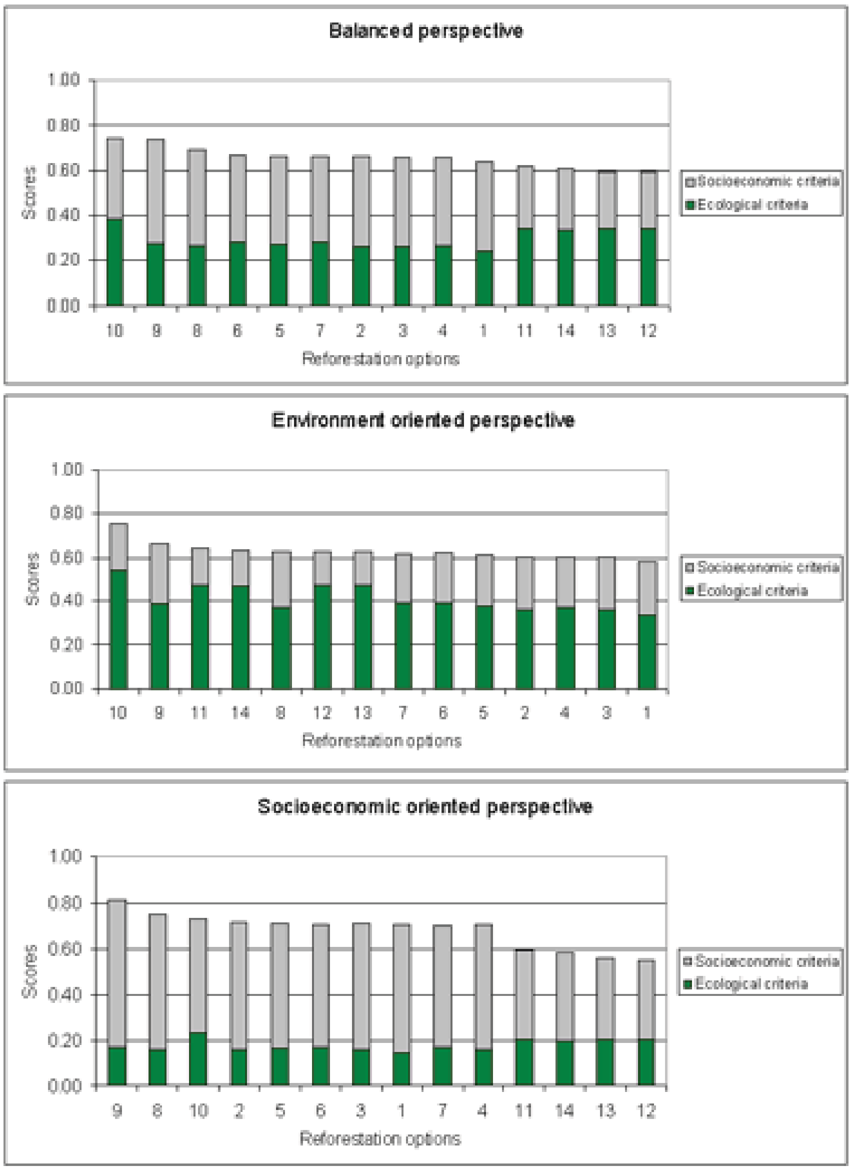

Figure 9.5 The rankings of reforestation options according to three evaluation perspectives: balanced (equal weights for ecological and socioeconomic criteria), environment oriented (greater weights for ecological criteria), and socioeconomic oriented (greater weights for socioeconomic criteria). The contribution of ecological and socioeconomic criteria is shown in green and grey, respectively. Adapted from Orsi and Geneletti 2010.

Table 9.5 Weights assigned to the criteria and groups of criteria adopted for comparison of reforestation options. While weights assigned to criteria are constant, those assigned to the groups vary according to three different evaluation perspectives: balanced (equal weights for ecological and socioeconomic criteria), environment oriented (greater weights for the ecological criteria), and socioeconomic oriented (greater weights for the socioeconomic criteria).

Group of criteria	Criteria	Perspectives		
		Balanced	**Environment oriented**	**Socioeconomic oriented**
Ecological		0.5	0.7	0.3
	I	*0.333*	*0.333*	*0.333*
	II	*0.333*	*0.333*	*0.333*
	III	*0.333*	*0.333*	*0.333*
Socioeconomic		0.5	0.3	0.7
	IV	*0.333*	*0.333*	*0.333*
	V	*0.333*	*0.333*	*0.333*
	VI	*0.333*	*0.333*	*0.333*

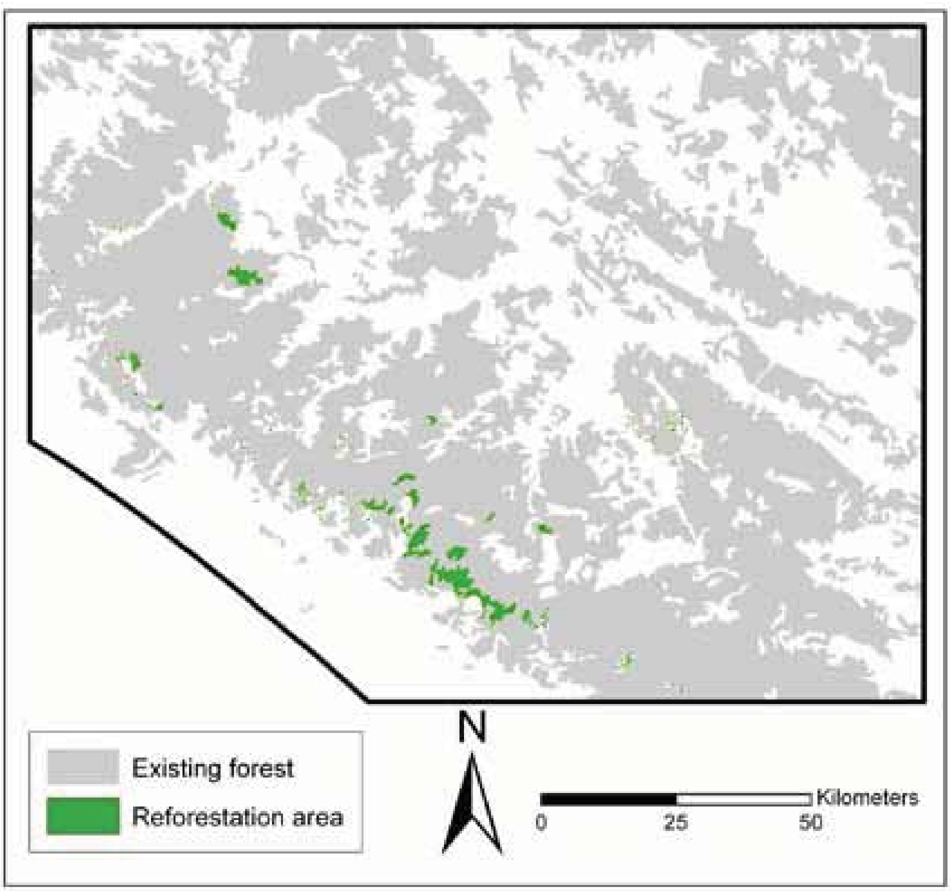

Figure 9.6 The most suitable reforestation option (option 10) was selected through a multicriteria evaluation that considered both ecological and socioeconomic variables. Adapted from Orsi and Geneletti 2010.

Finally, sensitivity analysis was conducted to test the robustness of the results with respect to changes in the evaluation inputs. Thousands of iterations were performed by changing value functions and weights within pre-selected uncertainty ranges. The results showed that these variations are not likely to affect the ranking of the 14 options (**Fig. 9.7**).

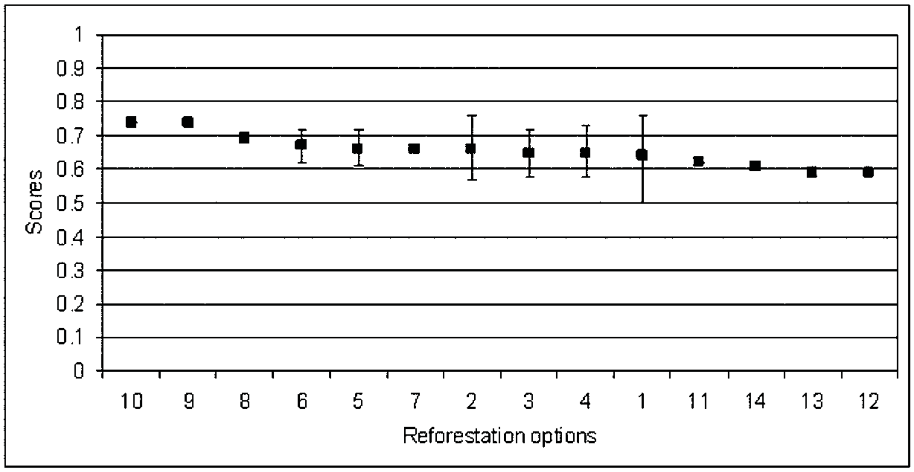

Figure 9.7 Results of the sensitivity analysis when uncertainty in the weights is considered. The black square represents the performance of a given reforestation option, while the bar represents the range within which the overall performance of that option varies depending on uncertainty in the weights. There is evidence of the stability of high-ranked and low-ranked options, whereas mid-ranked options show considerable variability. Adapted from Orsi and Geneletti (2010).

Lessons learned

- The proposed multicriteria approach provided a reliable tool to assess the suitability of land for forest restoration, to design restoration options and evaluate them according to their ability to promote biodiversity conservation. The methodology showed a convenient way to combine ecological and socioeconomic data, which are often characterized by different levels of spatial accuracy. The use of basic GIS operations and mathematical concepts made the approach user-friendly.
- The identification of restoration priorities might be considerably improved by applying spatial optimization techniques. These let the user define conditions that the restored landscape should fulfil (e.g. a given amount of restored forest at specific locations), and allocate land for restoration in a way that maximizes benefits to both nature and people (**Box 9.5**).
- A restoration plan can be successfully implemented when it is accepted by all the people (e.g. administrators, farmers, villagers) who have a stake in it, as discussed in the previous section. The proposed approach would take great advantage of involving stakeholders, and methodologies should be defined to account for their opinion and expectations. The last part of the approach (comparison phase), for example, might be transformed into a participatory phase where stakeholders comment on the maps of different options to identify the most acceptable ones.
- The lack of detailed information may result in the practical impossibility to use the proposed criteria. Hence, when selecting criteria and indicators it is useful to consider the possible alternative proxy measures (see Annex I and **Tables 9.1** and **9.2**). This is especially crucial in tropical regions, where considerable financial resources should be invested in data collection and preparation to compute the criteria that really describe the system at stake.

Box 9.5 A spatial optimization model for combining ecological and socioeconomic issues

F. Orsi, R.L. Church, D. Geneletti

The identification of forest restoration priorities involves several (e.g. hundreds or thousands) alternatives as each unit (and combination of units) of the landscape may be selectable. Decision problems of this kind are commonly defined as ‘continuous’ and have been successfully dealt with for years by applying spatial optimization techniques. These approaches aim to find the solution that either maximizes or minimizes an objective function subject to a set of constraints. Decision variables are generally assigned to each basic unit of analysis (e.g. raster cell) and may assume a binary value (1 or 0) depending on the decision taken (restoration or not). A spatial optimization model was applied to a study area located in central Chiapas, between the Sierra Madre mountain range on the south-western side and the Altos de Chiapas on the north-eastern side. The area has an overall size of about 430 km^2. Elevation ranges from 500 to 1400 m and the climate is classified as tropical wet and dry (Aw). Main forest types include: tropical dry, pine, oak and pine-oak. Humans live in several small villages across the area and rely on agriculture and cattle ranching. While these economic activities constitute the main cause of forest degradation and loss, forests are also exploited for timber and fuel wood. Forest restoration is needed to both protect biodiversity and sustain livelihoods. In particular, given that people are dependent on locally available natural resources, the restoration effort is likely to be sustainable if biodiversity is protected and local human communities are allowed to benefit from the ecosystem services that they have historically benefited from in an equally or more efficient way.

For the purpose of simplicity, forest restoration was considered just as reforestation. Four uses were considered: existing forest to be protected, existing forest to be harvested, reforested land to be protected, and reforested land to be harvested. The model attempts to assign a reforestation status to areas with the highest reforestation priority and to minimize the amount of existing forest given up for harvest. Several constraints ensure that: the timber demand is satisfied at each village location, timber sources are easily accessible from villages, a small portion of agricultural fields or pastures is converted to forest, a given amount of erosion-prone land is reforested and the reforestation budget is not exceeded. Higher reforestation priority was accorded to areas potentially supporting a high number of species, areas recently deforested and areas characterized by heavy fragmentation patterns. Reforestation costs were assumed to depend essentially on: the accessibility of sites, the plantation density, the natural regeneration potential and the current land use.

Considering US$2.5 million as the overall reforestation budget and 45 minutes as the acceptable travel time to reach forest stands from villages, the model identified an optimal solution implying the reforestation of about 3500 ha, of which nearly 60% was devoted to protection. The remaining part of the reforested area, plus an additional 200 ha of existing forest, were instead devoted to the satisfaction of villages’ timber demand. The model also allowed the exploration of tradeoffs between ecological benefits and the provision of ecosystem services. The amount of erosion-prone land to be stabilized through reforestation was increased step by step from the initial 1000 ha to 1800 ha and changes in the achievement of prioritization objectives (maximize the reforestation of species-rich areas, the reforestation of deforested areas, the reforestation of fragmented areas) and the use of existing forest observed.

The analysis showed that only when the erosion-related constraint is particularly demanding (1600–1800 ha) are the prioritization objectives poorly achieved, while the amount of existing forest to be given up for harvest is always quite low. Spatial optimization models offer considerable advantages, among which the possibility to pre-define conditions that restoration areas should fulfil, the opportunity to explore tradeoffs between different objectives and the warranty of finding the optimal solution seem particularly important in the practice.

The graph, below, shows the evolution of the landscape when the area of erosion-prone land to be stabilized through reforestation is increased from 1000 to 1800 ha. The graph (**Fig. 1**) shows that only when the constraint on erosion is particularly demanding (1600–1800 ha), does the ecological performance of the landscape significantly decrease, thus emphasizing that it is possible to both enhance the provision of ecosystem services and protect biodiversity.

Box 9.5 (cont.)

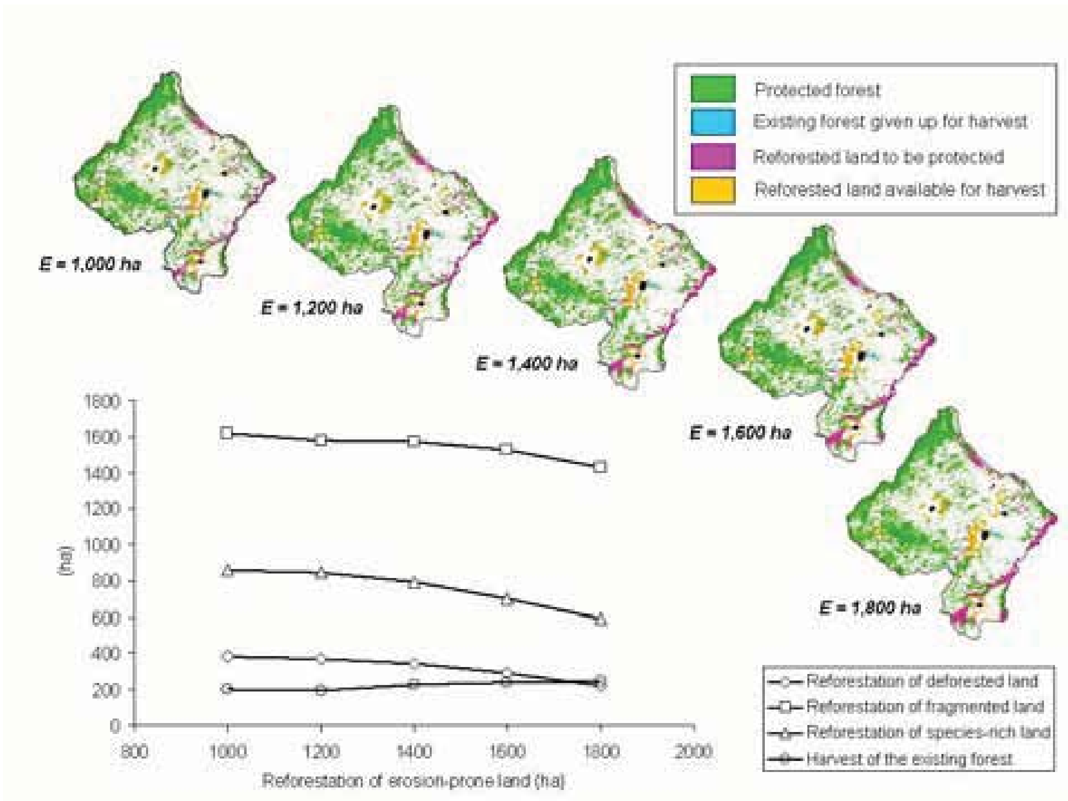

Figure 1 Use of a spatial optimization model for identifying restoration priorities.

Conclusions

The identification of forest restoration priorities involves considerable complexity. First of all, setting restoration priorities means clarifying the objectives of the restoration effort. Restoring forest for the provision of timber, for instance, is totally different from restoring it for the conservation of biodiversity or the stabilization of erosion-prone soils. Different objectives usually involve different priorities. That is why a sound analysis of such objectives is a necessary pre-condition to the success of any restoration plan. Once the objectives have been identified, decision analysis techniques may be applied to the actual definition of restoration sites. MCE techniques are particularly suitable for their ability to combine multiple decision criteria, incorporate the values of different stakeholders and deal with spatially-explicit information. However, the limits of MCE are many and should be handled carefully for proper decision making. The decision criteria, which are intimately connected to the above-mentioned restoration objectives, must be properly selected and adapted to a specific context, calling for the contribution of restoration experts. The evaluation of criteria is affected by uncertainty. Specific techniques (e.g. sensitivity analysis) may be applied to account for uncertainty factors. However, the final outcome should not be seen as the best solution, but rather as the most suitable option, in the light of the value judgements expressed by the stakeholders. Forest Landscape Restoration is carried out for people and with people, implying that the voice of local communities must be taken into account throughout the entire process.

Annexes

Annex 1 Criteria and indicators for the B factor, which refers to the need for biodiversity conservation. From Orsi *et al*. (2010).

Criteria	Indicators
Climatic conditions	Humidity; precipitation; temperature
Connectivity-corridors	Amount of interior habitat within a unit; corridor length; corridor width; distance from protected sites; linkages between habitat units; presence or absence of wild areas connected to the restoration area; types of linkages
Degree of threat	Area with threatened species; number of Red List species; presence or absence of Red List species; % of endangered forest; % of remained forest
Disturbance	Amount of area logged (ha); area of vegetation type after disturbance/area of vegetation type before disturbance; area/perimeter; density of steam crossings; distance from roads; disturbance classification; number of people depending upon the ecosystem; number of people living within the ecosystem; Natural Disturbance Type (NDT) classification; road density; % of agricultural area; % of area logged by slope class; % of invasive species; % of populated area
Diversity (ecosystem and landscape level)	Altitudinal variation; amount of dead wood; amount of deciduous trees; azimuthal variation; canopy cover; diversity of soil; landscape functional diversity; landscape structural diversity; presence or absence of diverse ecosystems at the landscape scale; presence or absence of water; quality of dead water
Diversity (species level)	Abundance; age; Beta diversity; evenness; Fisher's Alpha; forest density; number of birds; number of endemic species; number of interactions among species; number of keystone species; number of keystone species lost; number of major vegetation types; number of native species/number of exotic species; number of TER species; presence or absence of non-game species; Shannon diversity; species richness; % live/dead (mortality)
Diversity (genetic level)	Adaptive traits; canopy cover; genetic diversity among population; isozymes; number of stems per hectare by size class; neutral markers; nuclear inheritance; species-specific microsatellites
Ecosystem services	Carbon sequestration/productivity; distance from water; elevation; slope; soil retention (mass/ha); water provision (yield)
Fragmentation	Area of the fragments; core area; forest patch density; isolation; number of fragments; proximity; representativeness of the ecosystem in the world
Habitat availability	% ecosystem type by habitat; % type by watershed (500–5000 ha) (fine filter); % ecosystem type by habitat type by region (medium filter); % habitat type by region (coarse filter)
Historically forested area	Areas that were historically forested
Landscape degradation	Deforestation rate; fire frequency; frequency of landslides; land-use change (%); pollution indices; road density; soil erosion; volume of sediment-debris
Protected areas	Distance from protected areas; presence or absence of protected areas
Rarity	Presence or absence of rare species; representation of biotype in the broader landscape; uniqueness index
Recreation	Amenity value; number of people visiting the area; visual impact assessment
Remnants	Amount of primary and secondary forest at varying distances; distance from edge of forest; distance from forest of certain size; distance from remnant vegetation; distance from seed sources; presence or absence of adjacent areas with land-use types suitable for restoration; presence or absence of remnant vegetation; presence or absence of seed dispersers; tree and shrub density
Size	Area; area needed for restoring a vegetation type
Soil conditions	Nitrogen soil content; organic matter content of upper soil horizon; phosphorous soil content; soil macrofauna abundance; soil respiration; soil texture
Vegetation structure	Height distribution; horizontal structure: coarse woody debris-amount, size, level of decay; plant – strata diversity; structural stage; tree diameters; vertical structure: plant species composition, snags/wildlife trees-level of decay, cavity trees
Water ecosystem	Alkalinity; bank height; channel depth; channel width; dissolved O_2; distance from large rivers; hardness; length of water courses in the restoration areas; peak flow; pH; water clarity; wetness index; width of active floodplain

Annex 2 Criteria and indicators for the F factor, which refers to the feasibility of restoration interventions. From Orsi *et al.* (2010).

Criteria	Indicators
Ecological	
Accessibility	Distance from centres of appropriate capacities; distance from transport infrastructures; distance from cities; geomorphology; number of available vehicles; type of roads; type of vegetation
Climate	Climate change parameters; rainfall; relative humidity; wind
Degradation levels	Amount of old-growth trees; amount of remnant vegetation; amount of seed dispersers; compaction; erosion of topsoil; number of pioneer species; number of remnant tree species; nutrient depletion; soil fertility; species richness
Disturbance	Amount of herbivores; fire frequency; land use; livestock data; number of invasive species; people per Km^2; presence or absence of invasive species; presence or absence of noxious weeds; presence or absence of pests and diseases in the region; regeneration ability of invasive species; road density; type of livestock
Forest characteristics	Calliper – diameter; diversity; historical forest composition and structure; Landscape Biological Survey of Vegetetion (LaBiSV); number of exotic species; number of forest patches; number of stems per hectare by size class; patch distribution; presence or absence of desired plant species; presence or absence of mycorrhizae; presence or absence of old growth forest; presence or absence of secondary forest; species richness; tree height; uneven-aged/even aged forest; % live/dead; % threatened plants; % tree – plant species composition as a deviation from a baseline such as site series or late-seral plant community
Land-use conflicts	Differential land-cover use transformation rates; land use; landscape development plans; presence or absence of abandoned lands; presence or absence of private properties; presence or absence of utilities (power lines, etc.); suitability of land for alternative land uses; transformation matrix for each land-cover type
Natural regeneration potential	Distance from natural forest; distance from protected areas; distance to seed sources; growth potential; number of birds; number of seed trees and shrubs; pests and diseases adaptability; presence or absence of minimal biotic structures; presence or absence of biological corridors; presence or absence of unique genetic variants of populations using neutral markers, such as isozymes, microsatellites or DNA sequences; rhizomes and root material; seedling density; survival capacity; syndromes classification of the landscape unit; wind direction; % of species with different dispersal modes
Size of habitat	Area; number of fragments
Soil	Acidification of the substrate; altitude; aspect; bedrock type; bulk density; cation exchange capacity; compaction; concentrations of heavy metals; concentrations of pesticides; daily and annual temperature fluctuation; depth; erosion; fertility; microbial communities; organic matter (%); pH; plant-available phosphorous; precipitation; presence or absence of toxic chemicals; presence or absence of toxins; slope; slope below 35%; soil type; structure; total nitrogen
Water availability	Annual precipitation; aridity and humidity index; distance from rivers; elevation above the average groundwater level; field capacity of the soil; infiltration rate; precipitation distribution; soil depth

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10 DEVELOPMENT OF POLICY RECOMMENDATIONS AND MANAGEMENT STRATEGIES FOR RESTORATION OF DRYLAND FOREST LANDSCAPES

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Introduction

Forest landscape restoration typically involves the conciliation of interests of multiple stakeholders. As with other landscape-scale approaches, its ecological complexity spans several scales across both time and space (Levin, 1992; Young *et al.*, 2005; Cash *et al.*, 2006). Yet its practice and eventual success depends also on a number of complex human dimensions whose interactions develop over long periods of time (Higgs, 1997; Bradshaw, 2002; Naveh, 2005; Kanowski, 2010). It is probable that ecosystem goods and services provided by restored forests will eventually benefit not only local populations, but those located a considerable distance away (Buckley and Crone, 2008). On the other hand, someone has to pay for forest restoration, and rarely are local financial resources available over the time required to support restoration initiatives until signs of success are evident. As with most rural development actions, forest restoration projects usually require agreements on long-term use of consolidated land properties that involve local communities, grassroots groups, governmental agencies, urban social organizations, and others (Weiss, 2004). Forest landscape restoration should aim for legitimacy and avoid undemocratic approaches that lead to local and regional conflicts, which have limited the success of many conservation initiatives in developing countries (Lele *et al.*, 2010).

This chapter focuses on the development of public policy recommendations for forest landscape restoration, as this is probably the most crucial single issue that may affect the successful application and future practice of forest restoration in our study areas. As an example, Wuethrich (2007) describes a case where innovative public policies and coordinated governmental actions have made a major difference to conservation and restoration plans of Atlantic rainforest in the Rio de Janeiro state, Brazil. In the context of our study cases, the development and thoughtful use of decision-support tools, and the elaboration of local guidelines and management plans, all depend on, or provide input to the development of public policies. In most Latin American countries a common situation for forest restoration is to provide the common ground for the actions of a number of government agencies, whose actions relate to economic planning, social development, land tenure, agriculture, forestry, soil and water conservation, watershed management, and others. These agencies are prone to undertake contradictory actions. The intervention of multiple government interests is typically framed within a short-term period, usually only while some particular regulations and programmes promoted by temporary administrations take effect. On the other hand, desirable changes in rules and regulations to ensure a higher impact of restoration programmes may take years to be discussed and passed by local congresses. Conflicts may arise among stakeholder groups and local communities as some of the most valuable assets or commodities involved in forest restoration projects (particularly land) need to be committed for many years, despite the typically short life span of budgets. Therefore, a political pact has to be established that considers the current and long-term needs of all involved groups. It appears that to achieve success, forest landscape restoration has to be considered with respect to all the dimensions of a social construction.

National-level public policies on forest use and restoration often depend on the application of broad guidelines dictated by international development agencies, which themselves frequently pursue conflicting goals (e.g. United Nations, World Bank, Inter-American Development Bank). For example, it is considered that maximum total carbon sequestration may be attained in some cases with pine plantations (Fahey *et al*., 2010), which are being fostered as part of an overall policy to reduce the effects of greenhouse gases on climate change; yet it can be expected that the widespread establishment of such monospecific plantations will have severe detrimental effects on regional biodiversity (e.g. Richardson and Bond, 1991; González-Espinosa *et al.,* 2009; Galindo-Jaimes *et al.,* 2002; Richardson and Rejmánek, 2004). Grandia (2007) discusses how the support of the World Bank for a number of overlapping regional megaprojects contradicts biodiversity conservation in the Mesoamerican Biological Corridor initiative.

In this chapter we attempt to provide an integrated and synoptic view based on the research conducted in all the study areas of the ReForLan project in three countries: Argentina, Chile and Mexico. We contrast the biophysical attributes, the social, economic and cultural conditions, and the balance of land-use policies applied in recent decades in each region. From this qualitative analysis, we attempt to identify some basic emerging issues that could be considered in the development of public policy recommendations to conduct restoration programmes of dryland forest landscapes that employ decision-support tools, practical guidelines, and management plans. The research groups involved in the ReForLan project have widely different histories regarding their age and long-term research topics, producing diversity in their interactions with stakeholders in their respective social contexts.

The study areas and their scenarios for forest landscape restoration

Biophysical setting and land use

The range of environmental conditions encompassed by the study areas of the ReForLan project is enormous. In each area the particular gamut of environmental conditions has a major influence on the potential for forest landscape restoration. Although restricted to dryland forest ecosystems, considerable differences can be identified: patterns and processes of Mexican tropical dry forests have been contrasted with several types of Mediterranean vegetation in central Chile, Andean premontane subtropical habitats in norhwest Argentina and the ecotone of Austral Forest and Patagonian steppe in southern Argentina (Chapters 1–5, **Table 10.1**). The history of land use is also very diverse. The original forest cover in most areas has been severely reduced and agricultural land-use has been maintained for centuries, and even millennia in the Mesoamerican region (Carmack *et al*., 1996). Induced pastures and/or native rangelands devoted to cattle grazing (sometimes with sheep and goats) are a common feature in all regions, yet the practice of traditional indigenous agricultural systems (such as *milpa* or shifting agriculture, and highly diverse backyard orchards) only appears to remain in Mexico (principally in Oaxaca and to a lesser extent in central Chiapas).

Milpa **(maize field), Central Valley, Chiapas, Mexico.Photo: A. Martins**

Monitoring seedling for restoration in Veracruz. Photo: G. Myers

Table 10.1 Geographical, biophysical and land-use attributes of the study regions of ReForLan partners. UNCOMA = Universidad Nacional del Comahue (Bariloche, Argentina), FPY = Fundación ProYungas (Jujuy and Salta, Argentina), PUC = Pontificia Universidad Católica de Chile (Santiago, Chile), UACH = Universidad Austral de Chile (Valdivia, Chile), UC = Universidad de Concepción (Concepción, Chile), ECOSUR = El Colegio de la Frontera Sur (San Cristóbal de Las Casas, Chiapas, México), CIIDIR-IPN = Centro Interdisciplinario de Investigación para el Desarrollo Integral Regional-Instituto Politécnico Nacional (Oaxaca, Oaxaca, México), INECOL = Instituto de Ecología, A. C. (Xalapa, Veracruz, México), *MAT* = mean annual temperature, *MAR* = mean annual rainfall, AF = Austral Forest, APF = Andean Premontane Forest, CF = Chaco Forest, SF = Sclerophyllous Forest, MSF = Mediterranean Sclerophyllous Forest, MDDF= Mediterranean Deciduous Dry Forest, TDF = Tropical Dry Forest, SEPSF = Semi-evergreen Premontane Seasonal Forest, OF = Oak Forest, POF = Pine-Oak Forest, PF = Pine Forest, RP = Riparian Forest, G = Grasslands, SH = Shrublands. * Soil classification in central Chile and northwest Argentina follows the USDA-NRCS system; the FAO-UNESCO system is used in all other cases. HV = highly variable.

	Southern Argentina (UNCOMA)	**Central Chile (Coastal Range and Central Valley) (PUC, UACH, UC)**	**NW Argentina (FPY)**	**Central Chiapas, Mexico (ECOSUR)**	**Upper Mixteca, Oaxaca, Mexico (CIIDIR-IPN)**	**Central Veracruz, Mexico (INECOL)**
Latitude	39° 30'–43° 35'S	33° 30'–38° 00'S	22° 00'–24° 00'S	15° 50'– 17° 00'N	17° 00'– 18° 00'N	19° 17'–19° 25'N
Latitude (W)	71° 19'–72° 00'	71° 50'–72° 30'	63° 30'–65° 00'	92° 00'– 93° 30'	97° 00'– 98° 00'	96° 26'–96° 35'
Elevation (m)	800–1500	0–2260	350–750	500–1700	600–1500	40 –1100
MAT	12°	13°	21°	19°–26°	16°–18°	24°–26°
MAR	1500	100–500	900	800–1200	550–900	900
Predominant landforms	Steep and gentle hillsides and rolling plains	Steep hillsides on coastal range and gentle slopes and rolling plains between two cordillera systems	Steep and gentle hillsides and rolling plains	Steep and gentle hillsides and rolling plains	Steep and gentle hillsides and rolling plains	Gentle hillsides and rolling plains
Predominant soil types *	Andosols	Andisols, entisols, inceptisols	Entisols, inceptisols, mollisols	Lithosols, rendzines, luvisols, and regosols on limestone	Andosols, lithosols on limestone	Haplic phaeozems, lithosols, pelic vertisols
Major vegetation types	AF- steppe ecotone	SF, MSF, MDDF, RF, G, SH	APF, CF, RF	TDF, SEPSF, OF, POF, PF, RF	TDF, OF, POF, PF	TDF, RF

Table 10.1 (cont.)

	Southern Argentina (UNCOMA)	**Central Chile (Coastal Range and Central Valley) (PUC, UACH, UC)**	**NW Argentina (FPY)**	**Central Chiapas, Mexico (ECOSUR)**	**Upper Mixteca, Oaxaca, Mexico (CIIDIR-IPN)**	**Central Veracruz, Mexico (INECOL)**
Agricultural cover or main crops	Pastures and rangelands for cattle and sheep grazing	Vineyards, pastures, fruit-growing, livestock grazing	Irrigated sugarcane, soybean, pastures for cattle grazing	Induced pastures for cattle grazing, traditional annual crops, rain-fed crops with agrochemicals use, irrigated crops, fruit orchards, shade grown coffee plantations	Traditional annual crops, rangelands for sheep and goat grazing	Irrigated sugarcane, rain-fed annual crops, pastures, some fruit orchards
Major forest uses	Exotic tree plantations	Exotic tree plantations, firewood, charcoal	Firewood and timber from native trees	Firewood and timber from native trees, silvopastoral systems	Firewood and timber from native trees	Firewood and timber from native trees
Dependence on firewood	Low	High, HV	HV, low in cities, high in rural areas	High, HV	High	Low, HV
Use of non-timber forest products	Few	Several	Few	Many	Many	Many
Frequency and intensity of wildfires	Low/low	Medium/medium	Low/low	High/high	High/high	Medium/medium

The use of forest products is well embedded in the cultures of all study areas, and firewood from native tree species is still commonly used in many rural households; charcoal is produced and sold in many communities and in many instances may represent a significant source of income. The use of non-timber forest products is highly variable among the study areas. With the exception of the two study areas in Argentina, which report only a few cases, the traditional use of non-timber forest products in Chile and Mexico is quite common. In addition to timber and firewood, trees and forests may provide forage, medicines, fruits and seeds, edible fungi, pollen and nectar for honeybees, ornamental and ceremonial uses, and others (see Chapter 5; de Groot *et al.*, 2002; Marshall *et al.*, 2006). In Chile and Argentina, in addition to logging for timber and firewood, the remaining and mostly impoverished forest stands also face the threat of being replaced by commercial plantations of exotic trees. This is not yet the case in the Mexican study areas, but current and ambitious official restoration programmes hardly consider more than a few tree species, including both native aggressive and fast-growing species (such as pines) and exotics that are associated with highly reduced biodiversity (Carabias *et al.*, 2007). Commercial mechanized agriculture that uses high doses of agrochemicals is well established in Chile, Veracruz, and central Chiapas, and is rapidly expanding in northwest Argentina. Fire use is common to burn agricultural residues and old pastures, yet in highly variable regimes, destructive wildfires are more frequent in Chiapas and Oaxaca than in the other study areas.

Socioeconomic scenarios

The history of pre-Columbian population settlements and the resulting settlement patterns after their contact with colonizing European groups contrast markedly among the study areas (Mann, 2005). The history of movements and prevalence of ethnic groups within a given region may hold relevance for forest restoration, mostly because of the varying persistence of indigenous practices related to their native forest resources. In a more general way, the ethnic composition of a local population with their ensuing cultural and current political characteristics will influence the definition and application of public policies and the development of outreach activities.

Population density is rather low and mostly concentrated in towns and cities in Argentina, where mixed Caucasian immigrants who arrived within a relatively recent period (late 19th and first half of the 20th century) represent most of the population. From this extreme, a variable proportion of *mestizo* population is integrated with European immigrants in central Chile, to a lesser extent in central Veracruz, and with indigenous groups in central Chiapas. The Upper Mixteca of Oaxaca (southern Mexico) is mostly populated by indigenous Mixtec people (**Table 10.2**). Indigenous groups in Chile and Mexico densely occupied the study areas for centuries before their contact with European colonizers (Mann, 2005). It is noteworthy that most of the population in Oaxaca, and to a lesser extent in Chiapas, shows a highly scattered pattern, with families having their households close to their agricultural lands and maintaining free-range poultry and orchard trees in their 'backyards'; the uses of forest and grazing areas are communal and depend on agreements constructed by the whole group of landowners. A *mestizo* and relatively modern entrepreneurial rancher life-style prevails in most of central Chiapas and central Veracruz (**Boxes 10.1** and **10.2**).

Workshop in the community of Mata Mateo, in Paso de Ovejas, Veracruz, Mexico. Photo: A. Suárez

Firewood next to a potter's kiln, Chiapas, Mexico. Photo: B. Ferguson

Table 10.2 Social, economical, and forest restoration (FR) attributes of the study regions of the different ReForLan project institutional partners. See Table 10.1 for institutional acronyms. A = available, N/A = non-available, HV = highly variable. * *Mestizo* literally refers to people of mixed indigenous and Caucasian (typically Spanish) ancestry; it is usually applied to any Mexican who is Hispanicized to some degree irrespective of their actual ancestry. ***Ejidos* are pieces of land granted to peasant communities that hold them collectively and use them for farming and extraction of natural resources on the basis of community agreements; members of *ejidos* live within a community in designated areas for their households and other pieces of land are assigned for individual cultivation or communal use.

	Southern Argentina (UNCOMA)	**Central Chile (Coastal Range and Central Valley) (PUC, UACH, UC)**	**NW Argentina (FPY)**	**Central Chiapas, Mexico (ECOSUR)**	**Upper Mixteca, Oaxaca, Mexico (CIIDIR-IPN)**	**Central Veracruz, Mexico (INECOL)**
Population density	Low	High	Low	Very high	High	Medium
Population dispersion	Concentrated in towns and cities	Concentrated in towns and cities	Concentrated in towns and cities	A few major towns and cities; little scattered	Scattered	Concentrated in towns and cities
Ethnic group	Mostly Caucasian, some mixed indigenous	Mostly *mestizo**, some Caucasian, some mixed indigenous	Mostly mixed Caucasian immigrants, several indigenous groups	Mostly *mestizo**, some indigenous Zoque	Mostly indigenous Mixtec, some *mestizo**	*Mestizo* *
Poverty line	Half above and half below	Mostly above	Half above and half below	Mostly above, HV but rarely below	Mostly below	Mostly above, HV
General education level	Mostly elementary, some higher	Elementary and higher	Elementary and higher	Mostly barely elementary, HV	Mostly barely elementary, HV	Elementary and higher, HV
Land tenure	Private property	Private and state property	Private property	*Ejidos*** and private property	*Ejidos*** and private property	*Ejidos*** and private property
Land property size	Large	Small to large	Medium to large	Small to medium	Very small	Small to medium
Credit lines	Yes, HV	Yes, HV	Yes, HV	Mostly N/A	Mostly N/A	Yes, HV
Vulnerability to global markets	High	Very high	Very high	Moderate to high	Moderate, HV	High, HV
Migration to cities or abroad	Medium to high	Low to high	Low to medium	Medium to high	Medium to high	High
Agricultural intensification	Low	Very high	Very high	Moderate, HV	Low (moderate in plains)	Moderate, HV
Sustainability of current land uses	Medium/poor	Medium/poor	Medium/poor	Poor	Poor	Medium/poor
Passive or active FR	Mostly passive	Mostly active	Mostly passive	Both, low	Mostly active	Both
Diverse products through FR	Low	Low	Low	Medium	Medium/high	Medium, HV

Box 10.1 Contribution of livelihoods analysis to the establishment of priorities on restoration of tropical dry forest: a case study in the Central Depression of Chiapas, Mexico

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We adopted the sustainable livelihoods (SL) conceptual framework with the aim of selecting potential restoration areas and intervention strategies based on criteria and needs of local stakeholders. This allowed us to assess family level resources and visualize their interactions, recognize underlying social structures and impinging policies that create vulnerability in peasant families, and sketch major trends in the local communities.

Local participatory workshops on SL were held following a protocol agreed upon by the research group members in addition to interviews with key informants, and recognition of local conditions along altitudinal gradients. We conducted rapid sampling in the field to calculate an index of forest conditions (Ochoa-Gaona *et al*., 2010). The activities were conducted at two *ejidos*, Ejido 20 de Noviembre (92º53'W, 16º32'N; 450 m) and Ocuilapa de Juárez (93°24'W, 16°51'N; 940 m), both within the Central Depression region of Chiapas, southern Mexico. (*Ejido* refers to a particular land tenure concept that emerged from the Mexican Agrarian Reform conducted during most of the last century. *Ejidos* are communities where two land property rights coexist, private agricultural and household parcels and commons, usually forests and rangelands.) The study region has a tradition in the utilization and management of their natural resources spanning over four centuries, with a higher intensity of use observed during the last 60 years.

Analysis of SL indicates that these *ejidos* have a number differences between them: ranching prevails at Ejido 20 de Noviembre while a farmer-proletarian lifestyle prevails at Ocuilapa. Ranchers at Ejido 20 de Noviembre practice non-sustainable use of their natural resources, conserving forests in only 10% of their lands under agreed communitarian land tenure. Notwithstanding the observed intensification of their productive activities ranchers obtain meagre yields; currently, their main sources of income are remittances, paid work, and subsidies from the government, making them highly dependent and vulnerable. Their living standards are above the poverty limits regarding food, capability for changes, and assets. Population growth has stabilized and their fallow lands are under passive restoration (**Fig. 1**).

Peasants at Ocuilapa appear to practice a more sustainable use of their natural resources; by following a diversified strategy they have been able to preserve up to 45% of their forest area. Their major income sources are in agricultural activities that render them less vulnerable and dependent on external resources. They exceed reference poverty thresholds but obtain lower cash income than ranchers. A high proportion of their forested areas include shade-grown coffee plantations under the canopies of native tree species. Although these plantations had the lowest values when their ecological conditions were evaluated, they showed diversity values similar to tropical semi-evergreen forests. Internal trends and external pressures were evidenced and may put at risk the obtained sustainability. The local population continues to grow and craftwork activities are highly dependent on firewood and timber (pottery and carpentry, respectively) causing increasing direct pressure on forest cover. Even when landowners at Ocuilapa mostly regard themselves as coffee-growers they actually obtain most of their agricultural income from extensive cattle raising (**Fig. 1**).

The main issues, interests, objectives and criteria for forest restoration in the studied *ejidos* include:

- Landowners in both *ejidos* considered their major problems to be related to poverty and lack of governmental support for their productive activities. Ranchers also mentioned fire control while peasants cited lack of employment and technical options for agricultural systems.

Box 10.1 (cont.)

- Landowners in both *ejidos* would be willing to initiate a forest restoration project aiming at: productive development (agroforestry systems, timber and non-timber products), recovery of forests and degraded lands, and soil and water conservation. Peasants at Ocuilapa also mentioned that "forest restoration products should have a market".
- Criteria to be followed by ranchers in selecting a site for forest restoration would include that the project should be conducted within common use lands, reforesting areas within a reserve, and that restored lands should receive protection. On the other hand, peasants at Ocuilapa insisted in using ”lands most appropriate for the plantation of the restoration project … lands where other productive activities could not be affected … lands that otherwise were not productive". Finally, they mentioned that the chosen site for forest restoration could be useful ”to instil an environmental culture".

It is noteworthy that in both *ejidos* landowners have already identified those pieces of land in which they would be willing to conduct a forest restoration project. In both *ejidos* the plots are flat areas with extremely degraded soils that will require different restoration practices.

As a result of the workshop held at Ejido 20 de Noviembre the following priorities were identified: (1) to encourage ecoturism as a way to continue conservation activities in protected areas which at the same time will reduce pressure caused by further land division in the communal reserve, (2) to recover at Cerro Verde (as the community reserve is named) forest cover damaged by fires, and (3) to avoid human-caused fires. At Ocuilapa landowners highlighted their willingness to conduct better practices and receive technical guidance on novel agricultural systems.

Our results illustrate the local scale complexity and clearly indicate those sustainable rural livelihood elements that are already available and that could be used as a starting point to launch a restoration initiative with high potential success. It is relevant to consider the objectives, interests, and criteria that could move landowners to embrace such an initiative without ignoring those elements of their rural livelihoods that would be altered or modified, including a new organization of the strategic activities in each *ejido*, the change or strengthening of their assets, and the consequences for the living and employment conditions of their families.

These findings help to explain why environmental and rural development public policies are doomed to failure if they do not consider the needs, interests, and knowledge of local stakeholders.

Box 10.1 (cont.)

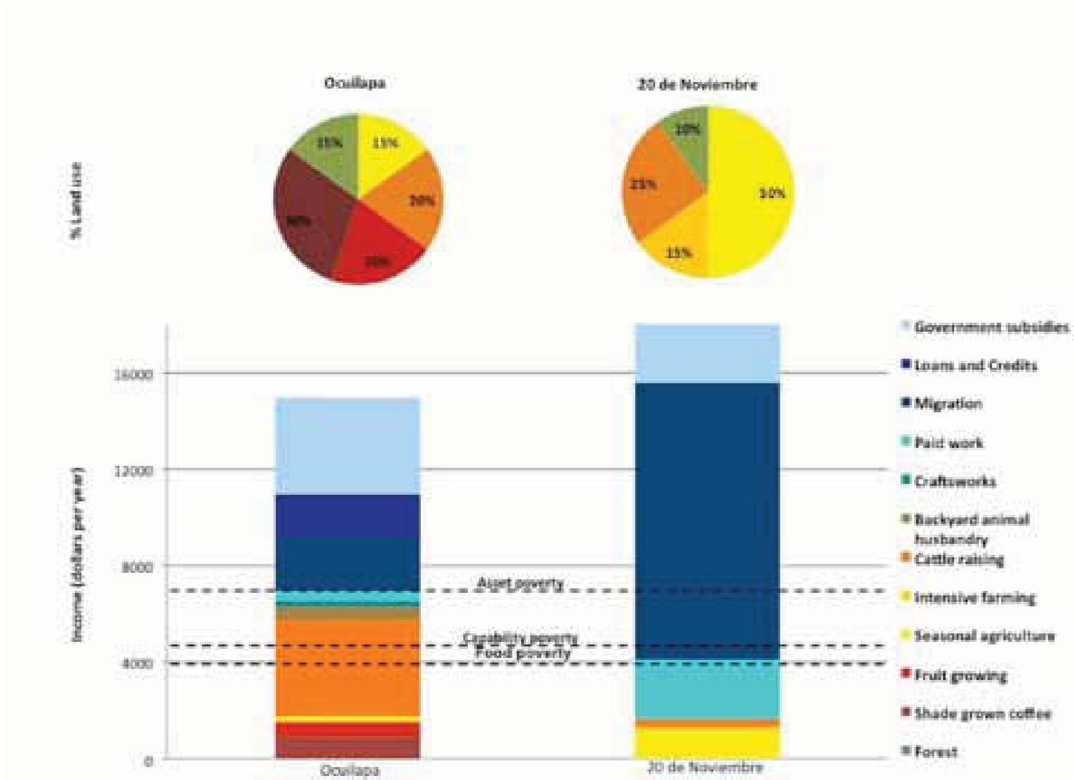


Figure 1 Pie charts showing percent area of each land-use type in the territory of each *ejido* (community). The bars indicate the composition of annual income in a 'typical family' derived from different activities and their relation to standardized poverty thresholds (CONEVAL, 2007). Different colours of the bar layers represent activities and land-use types; their thickness indicates contribution to income. The reference scale measuring poverty uses money amounts to account for poverty thresholds as defined by the National Council for the Evaluation of Social Development Policy (*Consejo Nacional de Evaluación de la Política de Desarrollo Social, CONEVAL*), which considers as a base value the cost of minimum foodstuffs included in the rural 'market basket'. Dashed lines refer to additive upper monetary limits for each poverty category: 'food or nourishing poverty' indicates a reference value related to the capability for food acquisition; 'capability poverty' refers to a limit to acquire health and education in addition to nourishment; 'asset poverty' indicates a monetary limit needed to cover nourishment, health, and education, as well as clothing, housing, and transportation. As a 'typical family' we considered a family group with seven members. The figures were prepared with original data obtained in a number of workshops on rural livelihoods conducted in the study area.

Box 10.2 Assessing the value of dry forests in two communities of the Central Depression of Chiapas, Mexico

J.A. Jiménez-Fernández, M.H. Huerta-Silva, N. Ramírez-Marcial

Mexico's forests have high environmental, social and economic value. About 8000 *ejidos* and agrarian communities hold about 80% of the country's forest lands, which for centuries have supported the survival and quality of human life. Forests provide food, fuel, timber, and other products for subsistence and commercial purposes. In addition, they play key roles in stabilizing ecosystems harbouring much biological diversity, helping to maintain air quality, water supply and soil conservation, and contributing to regulation of the Earth's temperature. There are no absolute forest values, as these depend on how forests are perceived by humans. These perceptions are dynamic and continuously changing. Value can be defined as the relative importance or the meaning that people in a particular locality assign to an asset within a given socioenvironmental and cultural context.

Through the use of semi-structured interviews aimed at landholders (*ejidatarios*) we estimated the value of dry forests in two locations of the Central Depression of Chiapas, in southern Mexico. The results indicate that over 80% of interviewed people in both locations retain some portion of their lands to provide forest goods and services, provided they have sufficient land. People with scarce land devote most or all of it to only one agricultural activity (rain-fed maize cropping). Yet even without conservation areas they recognize the importance of having trees in the plots to provide forest goods and services. People who maintain a portion of forest within their lands consider that they have a reserve of timber, in addition to other benefits that can be obtained directly from forests such as firewood, poles for fencing, timber to build their houses, medicines, organic fertilizers, or wildlife habitat for hunting. Some other indirect benefits from forests identified by landowners included: climate regulation, soil conservation, and protection against pests and diseases. To a lesser extent (5%) they recognized other universal values of forests as a source of life, happiness, tranquillity and satisfaction.

We used a methodology for analysis of rural livelihoods to determine the willingness of landowners in both locations to become involved in forest restoration projects. In the composition of rural livelihoods structures, both external and internal rules restrict the activities of family groups. There are events linked to vulnerability/stress, trends and internal and external policies limiting local action, on which in some cases stakeholders may have a direct action, but not always, as the actions may be imposed from outside. From this analysis we found that in one location, Ejido 20 de Noviembre, landholders have a more entrepreneurial rural profile with an ongoing urban transition; they are landowners with a productive/conservationist attitude, self-managed and with training for innovation. In the other locality, Ocuilapa de Juárez, the predominant livelihood is that of a forest peasant with an interest in plantations, craftworks, and farming; they currently are within a rural-to-urban transition, and maintain a conservationist attitude through a diversified approach to resource use. These peasants show a higher level of organization that empowers them to conduct their self-managed conservation and development projects.

Forest-use intensity varies with local needs and the availability of financial resources. For example, fuelwood extraction is performed weekly, but clearing for fencing posts is done once every year; timber is harvested any time when required to repair or build a house or shed, but timber for furniture is obtained on a weekly basis, and the collection of organic fertilizer from the forest floor is performed annually. The tree species most used at Ejido 20 de Noviembre were: *Leucaena diversifolia*, *Haematoxylum campechianum*, *Acacia pennatula* and *Cordia alliodora*. In Ocuilapa the most used species were *Acacia pennatula*, *Nectandra* spp. *Matayba oppositifolia*, *Mosquitoxylum jamaicense*, *Sideroxylon persimile*, *Cedrela odorata*, *Tabebuia rosea*, *Cordia alliodora* and *Trichospermum mexicanum*. The results of our study indicate that it is feasible to include these species in novel sustainable management schemes that involve community participation.

Poverty is widespread in all study areas, but the proportion of landowners that stand above or below standard poverty thresholds is highly variable. However, in no case do poor people account for less than half of the local population; more cases of extreme poverty were reported from Oaxaca (**Table 10.2**). Formal education levels are to some extent inversely related to poverty, and may have direct implications for the strategies to implement local restoration plans.

A private property land tenure regime dominates in the South American study areas, which may render agreements to coalesce lands for forest restoration a matter of personal decision. The size of land properties there tends to be medium or large, tending to hundreds or even thousands of hectares. A different situation is found in Mexico, where as a result of a long agrarian reform, private properties are only medium or small (in *ejidos* frequently extremely small), and decisions regarding forest use and restoration are made through community agreements (**Table 10.2**). Land tenure type may influence the possibility of obtaining credit lines to develop agricultural intensification and enter into global markets with export crops and other products, which can have conflicting direct effects on remaining forest areas and the restoration potential of degraded lands. Even when compared with the higher environmental risks that an agricultural intensification might create because of high input of agrochemicals and machinery use, it is evident that most currently practised land-use systems render a rather low value whenever it has been possible to assess their sustainability criteria.

Forest restoration practices are currently conducted to some extent in all study areas; yet the diversity of forest products obtained from restored stands appears to be lower than in mid- or late successional natural forests. Passive restoration predominates in fallow fields and in secondary stands where grazing and logging is regulated for a number of years. Active restoration over considerable areas is mostly performed with the introduction of fast growing species (pines, cedars), including exotics such as eucalypts (Lara *et al.*, 2003; Carabias *et al.*, 2007; Altamirano and Lara, 2010).

Legal and regulatory frameworks for public policy

The complexity of ecological restoration involves interactions among physical, biological, economical, social, cultural, and eventually, political elements (Weiss, 2004). In all study areas covered by the ReForLan project an overall national-level legal framework is available that aims to ensure sustainable use of forest resources. However, considerable differences exist among the underlying philosophies, scope, aims and details of the legal frameworks available in each country, as well as in the potential intervention of academic groups and organizations in transforming them (**Table 10.3**).

Table 10. 3 Legal and regulatory frameworks and participation in definition and (or) implementation of public policies (PP) on forest landscape restoration (FLR) by the different ReForLan project institutional partners in their respective study regions. See Table 10.1 for institutional acronyms. A = available, N/A = non-available, HV = highly variable, N = nationwide, S/P = state/province level, L/M = local/municipal scale.

	Southern Argentina (UNCOMA)	Central Chile (Coastal Range and Central Valley) (PUC, UACH, UC)	NW Argentina (FPY)	Central Chiapas, Mexico (ECOSUR)	Upper Mixteca, Oaxaca, Mexico (CIIDIR-IPN)	Central Veracruz, Mexico (INECOL)
PP in favour of FR	Yes (some)	Yes	Yes (some)	No	Yes	No
Some PP unfavourable to FR	Yes	Yes	Yes	Yes	Yes	Yes
Application of PP	Top-down	Markedly top-down	Top-down	Markedly top-down	Markedly top-down	Markedly top-down
Overlap of agencies acting on PP	Yes, high	Yes	Yes	Yes, high	Yes, high	Yes, high
Laws/regulations with sustainability criteria	A, N	A, N	A, N, S/P	A, N, S/P, L/M	A, N, S/P, L/M	A, N, S/P, L/M
Previous planning efforts and results, even if not directly related to FLR	A, N	A, N, S/P	A	A, N, S/P, L/M	A, N	A, N
Stakeholders participate in the design and implementation of PP on FLR	Yes, N	Yes, N, S/P, L/M	Yes, N, S/P, L/M	Yes, S/P, L/M	Yes, S/P, L/M	Yes, S/P, L/M
Stakeholders participate in major PP decisions on FLR	Yes, N	Yes, N, S/P, L/M	Yes (?)	No	Yes (?), L/M	Yes, S/P, L/M
Grassroots groups represented	No	Yes (some)	Yes (some)	Yes, L/M	Yes, L/M	Yes, L/M
Grassroots groups implement FLR	Yes	Yes (some)	No	No (a few)	No	No
Community-wide interest in FLR	Low	Low, HV	Low	Yes, HV	Yes, HV	HV
Territorial planning directly addressing FLR	No (former legislation did)	No	Yes	No, but other S/P planning A	No	No

Table 10.3 (cont.)

	Southern Argentina (UNCOMA)	**Central Chile (Coastal Range and Central Valley) (PUC, UACH, UC)**	**NW Argentina (FPY)**	**Central Chiapas, Mexico (ECOSUR)**	**Upper Mixteca, Oaxaca, Mexico (CIIDIR-IPN)**	**Central Veracruz, Mexico (INECOL)**
Partner/stakeholder interactions	Yes	Yes	Yes	Yes	Yes	Yes
Partner/stakeholder products	Yes, regional diagnosis, field guides	Yes, regional diagnosis, field guides	Yes, regional diagnosis, field guides, regulatory map	Yes, regional diagnosis, field guides	Yes, regional diagnosis, field guides	Yes, regional diagnosis, field guides
Monitoring of FLR by communities	No	No	Yes (some)	No	No	No
Community-wide awareness of FLR	No	No, HV	No	No, HV	Yes, HV	Yes, HV
FLR already linked to payment for ecosystem services	No	No	No	No, but some communities willing to	Yes	Yes (?)

In Mexico, legislation efforts focusing on environmental restoration have a history spanning more than one hundred years (Cervantes *et al.*, 2008; Álvarez Icaza and Muñoz Piña, 2008). Yet it is widely acknowledged in many areas of the country that local law enforcement of forest uses still has a long way to go. In contrast, a second generation law that explicitly considers conservation and restoration issues of native forests was approved in Chile in 2008 (*Ley de Bosque Nativo*). In Argentina one of the ReForLan partner institutions (*Fundación ProYungas*) was instrumental in the passing of a province-level decree on forest land planning and management in 2008 (*Decreto del Plan de Ordenamiento Territorial Adaptativo para Áreas Boscosas de la Provincia de Jujuy*). Public policies derived from the available legal and regulatory frameworks vary in their emphasis on forest restoration when applied in different regions under current local political administrations. For example, in the whole of Mexico a nationwide law might be applicable that considers forest restoration highly; yet state congresses and local executive authorities may favour other short-term ordinances linked to public policies and subsidies that drive actions that undermine forest sustainability (e.g. financing of deforestation for pasture establishment and livestock raising in Chiapas and Veracruz, or the production of agricultural commodities for export in northwest Argentina and central Chile).

A number of pitfalls or limitations in the definition and implementation of policies have been observed in all study areas, which eventually interfere with the long-term adoption of a forest restoration initiative by grassroots groups and communities. Most notable is the top-down application of public policies that insist in not taking into consideration local and long-term needs, capacities, and aspirations, thus dooming governmental projects to failure. In addition to this conflict between exogenous (macroeconomic) and endogenous interests, another common problematic feature of the political scenarios refers to the typical overlap of authority of governmental agencies, in most cases causing contradictory or competing actions among themselves (the typical case of local politics versus public policy).

In all study cases there is a clearly identified need for public policies on forest restoration to consider all stakeholders. Different stakeholders participate in the design and implementation of public policy on forest restoration; yet not in all cases do they participate in the decision-making process. Grassroots groups and community authorities may be represented in committees and councils but rarely participate in the decision-making process leading to the implementation of forest restoration. The interest and awareness shown by groups of landowners in conducting restoration plans in their lands is highly variable, and monitoring of the restoration practices is only rarely conducted by them. Payment for ecosystem services has gained worldwide momentum in recent years as a mechanism to finance long-term forest sustainable actions while providing water, carbon sequestration, climate regulation, soil for food production, biodiversity and other services to local and distant societies (Brauman *et al.*, 2007; Asquith *et al.*, 2008; Mooney *et al.,* 2009; but see Lele *et al.*, 2010). Yet, even when local groups in the study areas have expressed their willingness to participate, linking payment for ecosystem services with restoration plans is still far from being implemented as a viable approach to supporting forest restoration (**Box 10.3**).

Milpa **(maize) production, Chiapas, Mexico. Photo: N. Tejedor**

Box 10.3 Hydrological services and environmental decision making in Latin America

R.H. Manson, J.M. Rey-Benayas and M. González-Espinosa

Ecosystem services are defined as the conditions and processes by which ecosystems, and the biodiversity that they comprise, support and insure human well-being (Daily *et al*., 1997). For most of human history, such services and the ecosystems providing them were very abundant and easily accessible, and therefore were considered of little or no economic value. Nevertheless, exponential growth in human populations and their consumption of natural resources in the last few centuries has resulted in serious declines in these services (MEA, 2005), which have forced mankind to confront the enormous costs related to their restoration or replacement (Constanza *et al*., 1997, NRC, 2004; Rey Benayas *et al*., 2009). Of particular concern are the important hydrological services provided by forests (Myers, 1997; Bruijnzeel, 2004; Brauman *et al*., 2007) since forest cover is in decline worldwide, particularly in tropical regions (Laurance, 1999; Adedire, 2002). Over half of the world's freshwater is already being consumed, and what remains is increasingly inaccessible and contaminated (Postel *et al*., 1996; Gleick, 2000). Furthermore, global climate change is expected to greatly exacerbate these problems (IPCC, 2007).

Tropical dry forests (TDF) have the potential to provide a number of important ecosystem services to surrounding communities (Maass *et al*., 2005; Lemons, 2006). Nevertheless, the challenges related to the quantification of these services and their translation into economic value for the consideration of decision makers and the development of sustainable production strategies are daunting (Kremen and Ostfeld, 2005; Wunder, 2007; Rodríguez *et al.,* 2008). A promising approach to dealing with such issues is that being utilized by the regional network ProAgua. This network was designed to strengthen research initiatives and human capital dedicated to the study of hydrological services from a watershed perspective, and to develop strategies for the management and restoration of the ecosystems that provide them under different climate-change scenarios with the goal of improving the livelihoods of local communities in the Latin America and Caribbean region. This three-year project supported by CYTED (the Latin American Science & Technology Development Programme, http://www.cyted.org) involves 20 interdisciplinary teams from 10 countries (**Fig. 1**). They are using comparisons between strategic watersheds selected in each participating country to develop databases and standardized protocols for the conservation and restoration of hydrological services, the identification of future research priorities, and the creation of synergies by merging the respective strengths of each collaborating group. In addition, ProAgua seeks to train future generations of scientists and decision makers capable of sustainably managing water resources in the context of climate change via student exchanges and courses, as well as dissemination activities performed by means of symposia, conferences, workshops, a project web page (http://www.redproagua.com/), and the publication of both scientific and non-technical publications. Science outreach, to the general public and, particularly, to decision makers is a keystone objective of ProAgua.

The interdisciplinary and multi-stakeholder approach of the ProAgua network could be very valuable in regions dominated by TDFs. These forests are characterized by low primary productivity owing to rates of potential evapotranspiration that surpass precipitation for much of the year (Barradas and Fanjul, 1985) and, consequently, suffer high rates of deforestation owing to the perceived lack of economic value of natural forest cover (Trejo and Dirzo, 2000; Steininger *et al*., 2001; Reynolds *et al*., 2007; but see boxes with examples from communities in central Chiapas, Mexico). As water scarcity also makes the biodiversity of this type of forest slow to recover from human perturbations (Lemons, 2006), the degradation or elimination of TDF is all too prominent in landscapes in many regions of the world (MEA, 2005). The few studies performed to date to quantify and value the ecosystem services provided by TDF (Maass *et al.,* 2005; MEA, 2005) suggest that local communities lose much more than they gain by eliminating these types of forests, and that policy makers should seek strategies for reversing the observed trends.

Box 10.3 (cont.)

Such strategies should include programmes that reward landowners who adopt sustainable management practices that conserve or restore these forests and the ecosystem services they provide to local communities (Chan *et al.,* 2006; Wunder, 2007). Taxes on products and services to mitigate damage to the environment are widely implemented in developed countries (e.g. taxes on motor oil to pay for later management and recycling of the used oil). Similarly, schemes for taxes on products and services to restore damaged ecosystems should be implemented. Another option is more generous: tax deductions on donations intended to conserve and restore ecosystems. If tax deductions were 100%, donations by companies and citizens would be cost-free for them. By providing a market value for such services, these programmes would eliminate the externalities distorting traditional economic markets and create incentives for conservation that are equal to or greater than the opportunity costs foregone by limiting land-use options (Naidoo and Adamowicz, 2006). Nevertheless, unlike other forested ecosystems where a focus on the creation of payment schemes for one particular ecosystem service may be sufficient to sustain conservation and restoration efforts, the relatively low primary productivity of TDF makes the focus on multiple services a virtual necessity. While there is considerable interest in the mapping of multiple ecosystem services (Chan *et al.,* 2006; Naidoo and Ricketts, 2006; see Chapter 11), many challenges remain including the creation of interdisciplinary teams needed to quantify and value such services, as well as environmental education campaigns that permit the public and private sectors to understand the complex web of ecological interactions assuring their well-being and thus enhancing their interest in participating in such programmes. Forest landscape restoration should be seen as a keystone of a greener economy and a source of green employment. Labour is often the most expensive part of any project; if most of this labour were covered by 'volunteers' restoration costs would be dramatically lower.

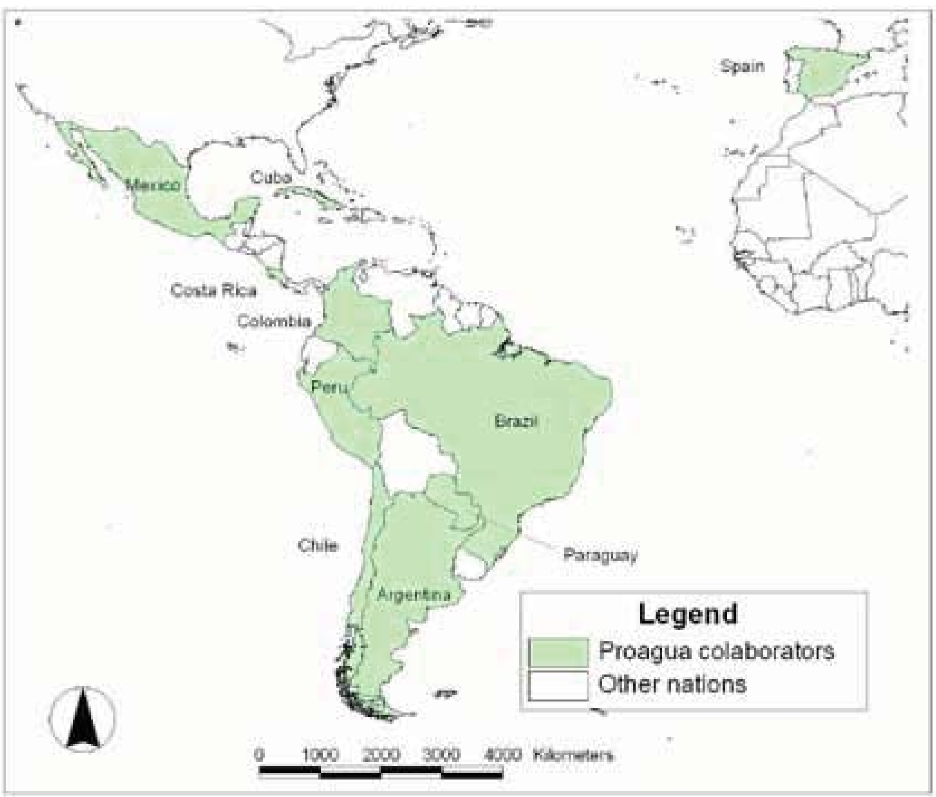


Figure 1 Map of the 10 countries currently comprising the ProAgua network focused on developing and disseminating strategies for conserving and restoring hydrological services in watersheds in Latin America and the Caribbean under different scenarios of climate change.

Lessons learned and suggestions for improving public policies

A comparative review of the major biophysical, social, economic, cultural, and political attributes found in the ReForLan study areas has been useful to attain an integrated and synoptic view of the prospects for restoration of dryland forest landscapes. Furthermore, a comparison among alternative solutions to common problems helps to pinpoint a number of issues to be considered in the design and implementation of improved public policies to support best practice in forest restoration. This approach is badly needed as public policies on rural development have been designed and implemented in the study regions for many years without achieving the anticipated success. Yet their repeated local failure obeys some general trends that emerge from our shared synoptic view; therefore, we take this opportunity to propose some actions that may prove helpful in solving critical problems. In the following paragraphs we suggest alternative actions as recommendations to help remedy some of the major obstacles facing forest restoration plans, and illustrate their current achievements and potential with a few selected case studies presented as separate boxes. For clarity, the recommendations are presented as groups of statements under separate subtitles for categories of activities, yet it is assumed that each would simultaneously influence multiple issues if they were ever implemented.

On land planning

Land planning and the elaboration of a basic and updated set of regional or local maps appear to be essential for both the definition and implementation of public policies. In our case studies the application of Geographic Information Systems (GIS) to generate maps has been particularly useful both for the overall development of a legal framework (the case of the Jujuy Province in northwest Argentina), as well as for supplying local maps used as input in workshops on sustainable rural livelihoods held with landowners in Chiapas and Oaxaca (see **Boxes 10.1**, **10.2** and **10.4**). This is not to say that GIS input in other study regions has not provided crucial products for the overall project, but it helps to emphasize the value of GIS-mediated interaction of the academic groups in supporting ongoing outreach activities and public policies.

Box 10.4 Using multicriteria decision-support tools in Rural Sustainable Development Councils in Chiapas, Mexico

M.R. Parra-Vázquez, M.H. Huerta-Silva, O.B. Herrera-Hernández, J.D. Golicher

Decision making within the context of environmental and rural development public policy is a complex issue owing to the multiple interactions among elements of the biophysical, economical and social subsystems. This is further complicated when decisions are made with the participation of government, and peasant and non-governmental organizations representing different interests, opinions and knowledge.

We have developed two initiatives to evaluate the possibilities for establishing coordination between environmental and rural development policies. Both initiatives were implemented within the Rural Sustainable Development Councils (*Consejos de Desarrollo Rural Sustentable*, *CODERS*): the first at a regional scale with the CODERS in the Central Highlands region of Chiapas; the second at the Villaflores municipality level in the Central Depression of Chiapas. CODERS are

Box 10.4 (cont.)

strategically positioned for decision making and generating public policies within the wider context of agricultural production systems; their institutional framework is built on participation of the social capital and its ability for strengthened decision making. Integration of local expert knowledge is considered crucial for decision making, and an approach based on sustainable development of land use was adopted in the proposal for better rural development policies aimed at poverty alleviation and environmental conservation.

The Highlands CODERS conducted a sustainability assessment of regional alternative agricultural practices based on criteria defined by Council members. A multicriteria Bayesian network tool was used to guide discussions and information structuring (**Fig. 1**). The results show that productive alternatives related to forest use (silviculture and shade-grown coffee) and traditional practices (*milpa* or shifting cultivation and backyard animal husbandry) had the highest sustainability ranking, while the lowest values were related to intensive cropping that makes use of abundant agrochemicals (commercial maize, horticulture, flower-growing) or extensive cattle ranching. This ranking was used to define strategic lines of action aimed at financing productive projects that would comply with restrictive guidelines, but also to look for new alternative systems or modifying the ones available.

The Highlands CODERS could restructure the funding allocated to agricultural activities, removing support to extensive cattle ranching in favour of agroecological projects. This change was implemented without governance conflicts as it was effected with an inclusive participatory planning scheme that considered all stakeholders involved in the decision-making process, where they actively participated and became aware about how relevant it may be to redeploy resources to productive alternatives.

On the other hand, CODERS at Villaflores started by conducting a number of workshops on rural livelihoods at the micro-regional scale from which a Municipal Agricultural Diagnosis was obtained. Afterwards, a forum for analysis of alternatives was organized in which both experts and grassroots groups participated. The Council assessed the sustainability of both current land-use systems and some novel ones that were of interest to peasants; criteria and indicators were built, and the resulting information was analyzed with the software DEFINIT, a multicriteria tool for decision making (**Fig. 2**).

The results indicated that current land-use systems (Alt1, Alt2 and Alt3) have very low sustainability when compared to alternative systems. CODERS concluded that public policies promoting extensive cattle ranching in central Chiapas rely on good funding and are antagonistic to an environmental conservation policy and represent a risk factor for conservation of forest areas. On the other hand, conservation policies operate with small budgets and can only favour a limited number of peasant communities.

From these two experiences we concluded:

- An overlap of authority exists among the different governmental offices involved in both CODERS, and such Councils provide an adequate space for their coordination.
- Expert knowledge should be taken into account in the decision-making process; this expert knowledge can be provided by specialists and decision makers, as well as peasants.
- Sustainability criteria agreed upon by Council members provide support and strengthen governance.
- Sustainability assessment in alternative productive systems and the use of decision-support tools proved helpful in (i) structuring thinking on available decisions, (ii) integrating and formalizing expert knowledge, (iii) obtaining a more thorough assessment of the available alternatives, (iv) visualizing the required information in a graphic way, and (v) making a more transparent decision. It also allowed the attainment of an interdisciplinary dialogue and consensus among Council members; finally, it helped to make explicit both the uncertainty and the information behind the decision-making process.

Box 10.4 (cont.)

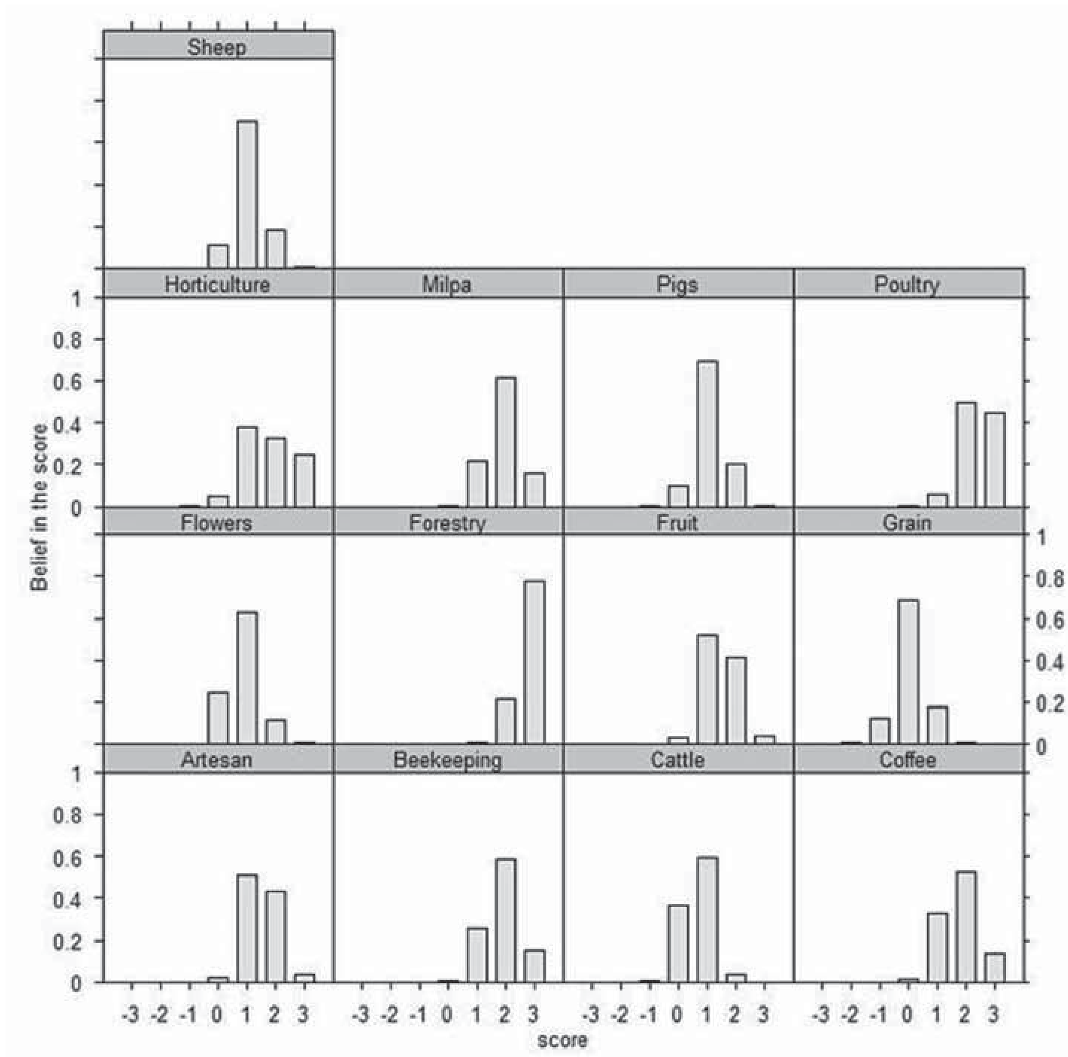


Figure 1 Probability of success of 13 agricultural production systems in the Highlands of Chiapas, Mexico. The distribution of bars indicates each system's probability of success according to the opinion of Council members. Each member assessed each production system on the basis of 24 indicators (sub-criteria) within six sustainability criteria. Values for each item ranged from −3 (poor sustainability) up to +3 (sustainable). The opinion of the whole group of Council members on all production systems was integrated using a Bayesian Belief Network. Non-timber forest products attained the highest value while the lowest was associated with conventional grain crops.

Box 10.4 (cont.)

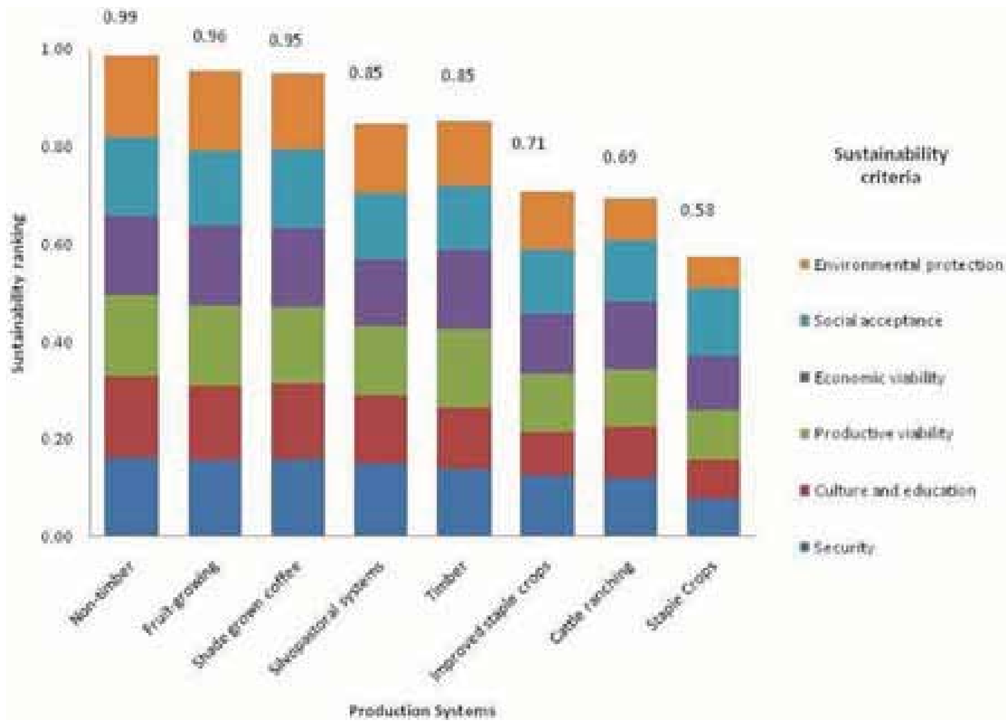


Figure 2 Sustainability levels in eight agricultural production systems at Sierra de Villaflores, Chiapas, Mexico. Sustainability was assessed on the basis of six sustainability criteria designed by members of CODERS. The three systems showing lowest sustainability values are the most widespread in the study region.

Collaborative land planning represents a powerful participatory strategy to motivate a deep-rooted and long-term interest among stakeholders. Analysis of rural livelihoods provides a knowledge base against which different models of public policy can be contrasted, evaluated, and tailored to intrinsically non-homogeneous communities. For example Lele *et al.* (2010) discuss how assumptions about homogeneity among communities have hindered progress of common approaches to biodiversity conservation such as the exclusionary protected natural area, and more recently the enterprise-based model and payments-based programmes. In our experience, rural livelihood analysis indicates that landowners have diverse needs and interests that should be considered as a whole and not segregated into major socioeconomic activities at the land-planning stage (e.g. agricultural development, poverty alleviation, forestry, biodiversity conservation, public health, education, infrastructure development), which can be identified in collective discussions (**Boxes 10.1**, **10.2**, **10.4** and **10.5**). Collaborative land-planning may provide an excellent opportunity for landowners to contribute to the planning process up to the decision-making stage, and it has been essential in allowing them to decide by themselves which tree species and which pieces of land will enter at different times into the restoration project. It is essential to work at an appropriate local or regional spatial scale where potential conflicts on sustainable use and restoration might be visualized and their solutions addressed. Forest restoration projects in some areas may be controversial among surrounding landholders because of implicit negative externalities, for example changing land use from commercial uses such as agriculture; yet efforts to control these externalities have the potential for off-site ecological benefits (Buckley and Crone, 2008).

Box 10.5 Lessons learned about social management in the Andean forests of Bolivia

X. Aramayo and B. Peredo

The Andean forests are found in the western part of Bolivia, over the Andes mountain range, between 700 and 4000 m in altitude. For the altitudinal range in which they are located, they have both high landscape and species diversity. Seven types of Andean ecoregions have been identified; of these the following forest types are the most outstanding: *Yungueños*, dry forest and *Puna* or high Andean forest. These forests are very important for the subsistence of the human population because they offer a series of environmental goods and services such as timber for construction, firewood for cooking and forage for livestock. They protect the soil from erosion and protect river basin headwaters, guaranteeing the provision of water for the surrounding communities and cities.

Approximately 60% of the population of Bolivia inhabits the Andean region, and because of this there is strong pressure on the Andean forests and other natural resources, which provokes their degradation and sometimes disappearance. For this reason the Swiss Intercooperation Foundation (*Fundación Suiza para el Desarrollo Técnico*), with the financial support of the Swiss Agency for Development and Cooperation (SDC), has encouraged and implemented, for some years now, in Bolivia, Ecuador and Peru, the conservation and sustainable management of the Andean forests. One of their most recent interventions has been the implementation of the *Programa Regional para la Gestión Social de Ecosistemas Forestales Andinos* (ECOBONA), managed at local, national and regional level, that applies policies, rules and instruments for the management of these ecosystems. This programme lasted three and a half years and was implemented in two areas of Bolivia: the province Ayopaya, in Cochabamba, and Mancomunidad of the Chuquisaca Centro.

Two issues were fundamental in carrying out the programme: to understand that the management of natural resources, and in particular forests, implicate the active participation of local public and private stakeholders, and that external companies must ensure that a facilitation role has been allowed, before the direct execution of the programme. This last issue must be accomplished in order to guarantee that actions remain sustainable after the institution retires from its role. Moreover, it is known that it is the stakeholders who must define the development avenues in their territory and act accordingly. In this sense, the facilitation companies (in the case of inter-cooperation) carry out a role in the support and assessment according to the needs of the people with which they work and according to the specific area of work.

In Bolivia ECOBONA presented five principal lines of action: (1) strengthening of capacity, (2) reduction of pressure on the forests through economic and political activities and regulatory processes, (3) ecological restoration, (4) management of forests, and (5) communication and sensitization. The implementation of the principal lines of action was flexible and depended on local capacities, the possibility to constitute alliances, the level of interest and motivation of the people involved, and the actual problems of the natural resources. However, in all cases the last three lines of action were fundamental and were applied in one way or another to make sure they were adhered to.

In the area of politics and regulation, the work was centred on the urge to elaborate local regulations for the management and conservation of natural resources. The regulations were made at municipal level, or at regional level (sub-municipal) when the territory was too large. The most interesting feature of the process was the wide participation of the representatives of the social organizations, who were in charge of the whole process after receiving training, and the municipal governments who finally approved the regulations by means of municipal ordinance. Local regulations have been well received because they were suggested by the people themselves and their level of application is growing, to the extent that they are now generalized. They are considered an adequate instrument for the management of the natural resources and for the conservation of forests. Even though the application of public policies, and in particular

Box 10.5 (cont.)

the regulations, correspond to the local people involved, the programme considered that it was necessary to provide support for a prudent period. Support was considered essential in meetings; to know the level of application, to analyze the problems found along the way and to see the possible ways of overcoming them.

A second area that was worked on was the ecological restoration and management of forest, according to the type of forest, to optimize the use and recuperation of degraded areas. For this, ECOBONA put emphasis on the following actions:

- The preservation of forest, understood as the strict protection of forest or forest found within an area of strategic character, particularly in high areas of watersheds and on slopes, owing to their importance in the provision of water. Enclosures were established to close off areas, for risk management, and fire control. Regulations were elaborated to impede the abuse or destruction of these forest areas.
- The recuperation of degraded areas by means of reforestation actions with both native and exotic species, especially in those areas completely denuded. This has been one of the principal requests of communities, particularly in high elevation Andean areas.
- The enrichment of forests with native forest species of high commercial value.
- The application of forestry management plans aimed at fulfilling traditional needs and purposes that may propose technical measures but have wide social acceptance. The addition of "with traditional needs and purposes" is to distinguish conventional management, whose intention or purpose is commercial, from the management plans of rural communities at Chuquisaca, whose management and use of forests is for self-sufficiency. In fact, the largest pressures on the Andean forests are the traditional uses (firewood, construction materials, tools, agriculture, fires, pastures) that are carried out with limited technical criteria.

Finally, a relevant aspect was the strengthening of capacities of the local groups involved, so that actions have continuity and remain in their own hands. Support was provided in a number of areas from institutional strengthening of the *alcaldías* to the training of forest management representatives of social organizations. Achievements so far obtained have been stimulating and have become more successful as time goes on; one example is that local people feel more identified and represented in the project. In all cases it is thought that the Andean forests are benefitting thanks to their use under the appropriate criteria and regulations.

In most cases, earlier desirable land-planning initiatives have been available but have previously not been fully implemented (e.g. management plans of extant nature reserves, several municipal and state plans, etc.). The resulting land planning should identify and prioritize areas where land-use intensification can be profitably effected, helping to release pressure on marginal lands where restoration of forest cover may be most needed and is most likely to add value (**Box 10.6**).

Box 10.6 Sustainable forest management in *Yungas* forests: a protocol to develop a forest management plan and implementation in an experimental farm

P.M. Eliano, C. Badinier, L.R. Malizia

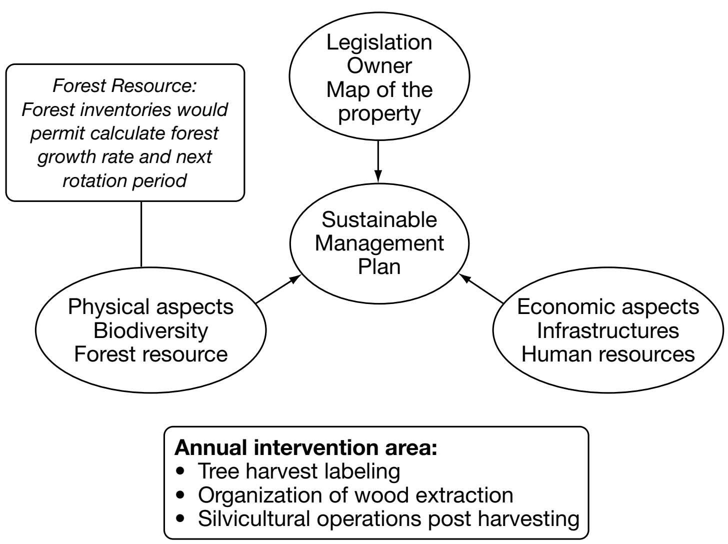
Yungas or mountain forests cover four million ha in northwestern Argentina. They represent a critical ecosystem in Argentina owing to their high species richness, high timber value and the provision of ecosystem services (water provision, soil protection). Over the past century, this ecosystem has been heavily logged for timber production and transformed into agriculture. We have (1) developed a theoretical protocol for the elaboration of sustainable forest management plans, and (2) implemented a practical example of a plan in a local forest farm within the area of interest, to promote sustainable forestry among the stakeholders of the forestry sector.

First, an analysis of the forestry sector in the region was conducted in order to detect its strengths and weaknesses. Among the strengths we identified strong demand for timber, adaptation capacity, increase in forest information, a new forest law that promotes sustainable use; weaknesses include: lack of knowledge on forest dynamics, poor organization of the forestry realm, lack of education and information, strong illegal markets, deforestation, and lack of public incentives. This preliminary analysis showed a strong need for improving forestry management practices compatible with the preservation of native forests. Also, it indicated a clear need for reforestation and restoration.

Second, meetings with the forest sector (enterprises, grassroots associations, universities, government agencies, NGOs) were organized to elaborate a protocol for a sustainable management plan to be implemented in private properties. It was important to involve all stakeholders in this participatory process.

Third, in order to evaluate the protocol, a pilot management plan was implemented in a 12,000 ha forest farm (**Fig. 1**). This action has been extremely important to prove the feasibility and effectiveness of the protocol and as a demonstrative site for stakeholders of the forestry sector.

Figure 1 Process for developing forest management plans in *Yungas*.



In conclusion, this work promotes the integration of forest production and conservation. Sustainable management plans can be implemented in north-west Argentina to restore native forest structure and diversity without penalizing timber production. However, this work also emphasizes the poor level of current knowledge on forest dynamics; this knowledge is vital to achieve sustainability in forest management. This work is the continuation of a long process aimed at improving forestry practices in *Yungas*.

On legal and regulatory frameworks

Legal instruments continue to evolve in Latin American countries. However, a number of issues that have been clearly identified and are scientifically well known have not yet entered into discussion and remain marginal to advances in legislation; for example, the lack of appreciation of the ecological concept of 'landscape', which remains in a legal limbo within the Mexican legislation owing to being mistakenly equated only with natural scenery (**Box 10.7**). Among these, the central importance of adopting sustainability as a key concept to prioritize development projects should be emphasized. Sustainability itself is a concept that continues to evolve, and it should therefore be further defined whenever needed to adapt it to changing conditions.

Box 10.7 Public policy and land-use change in central Veracruz (Mexico): an important link in efforts to restore a tropical dry forest landscape

J.A. Montero-Solano, R.H. Manson, F. López-Barrera

One of the main causes of tropical forest loss is conversion to other land uses. These conversions are to some extent determined by public policies (Geist and Lambin, 2002). Within Mexican forestry policy, dry deciduous tropical forest has lacked economic value. In the Mexican state of Veracruz this type of ecosystem, deemed "low forests of the plain", has not been thoroughly studied because of the "low value of its species" (de la Peña, 1946). Owing to their small diameter and reduced height, these trees were not considered attractive for logging; however, visible signs of degradation were still noted in many areas (de la Peña, 1946). Perhaps due to this perception, article 251 of the Federal Law of Agrarian Reform (1971) stated that agricultural or livestock property could not remain unexploited for more than two consecutive years, which encouraged even more conversion of forest cover in the region (Montero-Solano *et al*., unpublished).

An analysis of satellite imagery for the municipality of Paso de Ovejas in central Veracruz, shows that between 1990 and 2000 the area under pasture increased at the expense of areas occupied by both disturbed and preserved forest (**Table 1**), resulting in a high annual deforestation rate of 21% for undisturbed forest, and 6.58% for total forest cover (including secondary forests). This loss may be related to agricultural programmes such as PROCAMPO, which was initiated in 1995, or to PROGAN; these programmes encouraged deforestation by inducing landowners to convert forest into farmland (Merino Pérez, 2004).

From 2004 to 2008, forest programmes have supported the reforestation of only 377 ha or 1% of the municipality, even though this area is eligible to receive payments for biodiversity conservation through national programmes focused on the conservation and restoration of forest ecosystem services. Since 2000, migration out of the municipality to major Mexican urban centres and the United States has been notable (Méndez Main, unpublished) and is linked to the abandonment of land that could partially explain the recovery of the forest area between 2000 and 2007 (**Table 1**).

Although the Veracruz state government recognizes that the expansion of the agricultural and livestock frontier has been at the expense of forest resources, it is still encouraging this land-use change "in order to gradually convert the semi-arid zones of the state into areas of higher productivity" (Herrera Beltrán, 2005). This change is reflected in livestock production activity with pastures doubling in area between 1973 and 2007 (**Table 1**).

Box 10.7 (cont.)

Table 1 Percentage cover change of different land-use/cover classes at Paso de Ovejas, Veracruz, as obtained from satellite imagery from 1973 to 2007.

Land-use class	1973	1990	2000	2007
Forest	2.06	6.28	0.73	4.79
Secondary forest	25.76	25.10	15.52	22.35
Tree plantations	0.00	2.72	1.68	2.11
Grassland	21.85	8.04	32.29	43.92
Irrigated agriculture	22.18	21.37	18.93	13.15
Rain-fed agriculture	27.74	33.81	27.90	10.06
Urban	0.34	2.07	2.30	2.32
Other	0.07	0.62	0.65	1.30

The General Law of Ecological Equilibrium and Environmental Protection (LGEEPA) and the General Law on Sustainable Forestry Development (LGDFS) establish some limits on land-use changes and the elimination of forest cover in the landscape; yet these appear insufficient to counteract other public policies intended to have the opposite effect. Article 3, Section I of the Regulations of the LGEEPA, regarding environmental impacts, defines land-use change as ”modification of the natural or main vocation of the land, carried out by man through the partial or total removal of natural vegetation”. Although the LGDFS prohibits changes in forest cover, except under extraordinary circumstances, deforestation often occurs owing to unauthorized and often small-scale changes in land use (FAO, 2005a).

In legal terms, the concept of ’landscape’ is poorly defined in Mexico. In Article 7, section XXXVII of the LGDFS it is stated that the landscape is an environmental service. With reference to the use of forest resources, Article 100 of the LGDGS states that ”authorization will not be given if the proposed use could jeopardize the respective populations and environmental functions of the ecosystems, including soil, water and landscape”. Although the LGDFS does not define the concept of landscape and states that ’in the regulations and official Mexican standards issued for such purposes, the criteria, indicators, and corresponding measures are established’, the regulations of the LGDFS omit any reference to the landscape, and no official Mexican standard exists which determines the characteristics or parameters of a landscape. Such ambiguity leaves the concept of landscape in a state of legal limbo and limits the monitoring of this environmental service, which can include services of connectivity and wildlife habitat, the regulation of climate, scenic beauty, and cultural value (Maass *et al*., 2005).

Under the terms of the LGEEPA, the landscape is considered an element of ecological zoning, and is stipulated as such in Article 23 section II, paragraph f, such that public programmes of ecological zoning must incorporate a landscape component. Once again, however, a legal definition of the characteristics of the landscape is lacking.

Since a landscape is a geographic area in which relationships between human activity and the environment have created ecological, socioeconomic and cultural patterns and feedback mechanisms that contribute to the formation of perceptions and values, a strategy for institutional intervention through the ”social construction of a planning process and territorial regulations” (Aguilar Bellamy, 2006) is urgently required.

In summary, this review of the landscape of the municipality of Paso de Ovejas in the central portion of the state of Veracruz reveals that territorial regulation is required to identify areas for forest restoration (including passive, low-cost restoration), an activity that could be financed through the dual economic stimuli of payments for environmental services and investment in agroforestry systems and ecotourism projects. Such schemes must consider the history of public policies in the region, reversing the entrenched perception of the poor value of the low tropical forest and considering various policies and incentives for both large and small landowners in the municipality.

Other issues so far not considered in most applicable laws refer to technical constraints that should be observed by restoration programmes. Further discussion should be supported on the development and implementation of more advanced or sophisticated laws on obligations and practices relating to forest restoration by landowners. These laws and regulations should set minimum standards or numbers on: (i) tree species/ha to be planted in restoration programmes to ensure plant diversity, (ii) individual trees from which seed must be collected for utilization in restoration nurseries to ensure genetic variation, (iii) the timescale over which restoration plantations or passive restoration plots and enclosure fences should be maintained, (iv) the proportion of community or individually-owned land that should be maintained with forest cover, (v) use of living fences that might help to connect forested areas, and (vi) enforcement of establishment and maintenance of forest cover in critical areas such as river margins and high steep slopes. Regional networks of research institutions (see **Box 10.3** on the ProAgua network involving 10 Latin American countries) have the capacity to organize forums for a regional or worldwide systematic review and analysis of current laws and regulations applicable to forest sustainability and restoration, which would prove helpful in improving state and national legal frameworks.

On interagency relationships and application of regulations

Overlaps of the legal domain and derived policies available to different federal, state and municipal agencies have repeatedly been found. Although this condition has generally been observed to result in a cancellation or interference of efforts, it is true that it could be profitably turned around if different governmental agents interact at the same level in a mixed and well-regulated committee environment.

A relevant example in this context is provided by the Rural Sustainable Development Councils (*Consejos de Desarrollo Rural Sustentable*), which have been included in Mexican national legislation. By law these must include a number of representatives or delegates of the ministries deemed to be involved in rural development plans and actions (**Box 10.4**). At the municipality level these councils have full capacity as decision makers as they are plural and democratic entities with well-represented grassroots and technical groups, which can best prioritize local and regional plans. The application of decision-support tools such as Bayesian Belief Networks, spatial multicriteria analysis, and others, has been possible and particularly fruitful in some of these councils at the municipal level (**Box 10.4**). Yet, the dependence of their resolutions on higher political levels has impeded their decisions taking full effect during the implementation of the agreed programmes.

The application of plans for forest restoration should be conducted within a broad and transparent framework that ensures stakeholders are kept well-informed on the issues, proposed solutions and executive actions of the government agencies and officers. Finally, an effort should be made to alleviate the heavy bureaucratic load currently imposed on landholders when applying for incentives and funding. This requirement is usually centralized in large urban centres imposing transportation expenses on poor landholders; if needed this could be alleviated with mobile offices visiting the communities.

On economic issues

Success of public policies on forest restoration will eventually face the challenge of applying sound economic alternatives (**Boxes 10.4**, **10.6**, and **10.8**). Success of public policies will depend in the end on the application of effective policy instruments embedded in an explicit economic context (**Box 10.8**). Even if landowners are not situated under the extreme poverty line, only rarely will they have sufficient assets to finance their technological upgrading. Innovative financial mechanisms must be found to pay for the transition including the development of the required infrastructure facilities (Collins *et al.,* 2009; **Box 10.3**). Yet it should be remembered that poor landowners are exposed to regional and global mainstream market trends that condition their ability to benefit from public policies and distort the diversified structure of their productive systems.

Box 10.8 What's next? Design and implementation of policy instruments for forest restoration and management in Latin America

I. Schiappacasse, L. Nahuelhual

Throughout this book the authors have stressed the importance and threatened status of dryland forest ecosystems in Latin America. Taking into account these considerations, they have developed policy recommendations to support restoration of dryland forests. The next step is to 'translate' all this valuable information into effective policy instruments for the restoration of dryland forests. The overall goal of a policy framework should be to achieve efficient long-term restoration and sustainable use of biodiversity.

Sterner (2003) states that one frustration that usually has to be confronted is that seemingly simple solutions to serious environmental problems exist but are never implemented. Thus we need to design feasible policy instruments that can be successfully implemented. First of all, we must understand why environmental policy is needed in the context of restoration. The reasons are interlinked with the evolution of property rights:

1. **Market failure:** this is a technical term that broadly refers to conditions under which markets do not produce optimal welfare outcomes. Such failures include: (1) *external effects (externalities),* non-market side effects of production or consumption, such as soil erosion (negative externality) caused by unsuitable agricultural practices, or the hydrological services (positive externality) provided by a protected watershed. The problem is that these negative (positive) externalities are real costs (benefits) but these costs are not borne (perceived) by the individual that causes the damage (benefit). Restoration projects bring about social benefits (positive externalities) that are not accounted for in markets. As a result, too few restoration actions are undertaken; (2) *public goods*, products or services whose consumption is not excludable and non-rival. As a result, the market tends to under supply them since no one can be forbidden from consuming them. Faced with the provision of public goods people tend to understate their true willingness to pay, and therefore aggregated benefits are underestimated. Ecosystem services associated with restoration are public goods that, given their features, will not be provided privately. The solution is direct government provision.
2. **Policy failure:** Policies reflect economic interests, and in some cases, there may not be a single policy that is 'optimal' for every group in society. Policy failures arise when public intervention is needed but not undertaken or when interventions come to aggravate an already existing market failure. For example, subsidization of restoration and reforestation actions may induce landowners to clear existing native forest in order to receive the subsidy. This case has been documented by some authors in Costa Rica within the context of the payment for ecosystem services (PES) programmes.

We can distinguish two main types of policy instruments in the context of forest restoration and sustainable resource use: (1) non market-based instruments (i.e. establishment of protected areas, environmental standards) and (2) market-based instruments (economic regulations such as taxes

Box 10.8 (cont.)

and subsidies). Market-based instruments may be a more effective tool for the sustainable use of dryland forests than non-market ones because they explicitly address the causes or mechanisms that threaten forest ecosystems. Properly designed and implemented, economic instruments can become an important component of an incentive-based approach to restoration.

In recent years Latin America has experienced an incipient development of market-oriented policy instruments, particularly regarding PES in certain countries (e.g. Asquith *et al.,* 2008; Pagiola, 2008). However, by and large, environmental policy in Latin American countries continues to rely on direct regulation instruments whereas the development and use of market-based mechanisms is in its infancy. The challenge is to develop innovative policy instruments based on the alternatives provided by the environmental economics literature:

a. *Direct regulations*. This type of instrument refers to what economists call command-and-control regulations, including restrictions on access and land use, for instance. They offer an alternative way to achieve conservation and restoration objectives. This type of instrument is usually criticized because its lack of flexibility. A command-and-control regulation, for example, requiring that forest be restored under certain conditions (e.g. as a compensation mechanism for development projects that deplete some type of forest) would apply to all forests, irrespective of the level of benefits they provide or the cost of restoring them.

b. *Subsidies*. There are several forms of subsidies. For instance, from the perspective of PES recipients, the payment acts like an environmental subsidy aimed at inducing increases in environmentally beneficial activities. This kind of instrument has attracted increasing interest as a mechanism to translate external, non-market values of the environment into real financial incentives for local actors to provide environmental services. Unlike environmental taxes, however, environmental subsidies suffer from several sources of potential inefficiency, and thus are usually considered a second-best solution (Baumol and Oates, 1988).

c. *Taxes*. Theory tells us that environmental taxes (charges on environmentally-damaging activities) can, like environmental subsidies, help internalize the value of environmental services in private land-use decisions. Environmental taxes suffer from fewer of the problems presented above, so they might be considered superior to environmental subsidies and command-and-control regulations. Distributional concerns often militate against the use of environmental taxes, however (Engel *et al.,* 2008). For instance, taxes on agricultural activities undertaken in restoration priority areas would impose the cost of forest restoration on land users rather than on service users.

As we can conclude from this brief analysis, there is no 'perfect' or single policy instrument to promote restoration and new ones can be developed. However, the choice and design of particular instruments is complex and dependent upon specific institutional, economic and social needs. When market-based instruments are cost-effective they should be promoted. Nonetheless, in many cases it will also be necessary to use non-market-based instruments in an effective policy mix, in order to achieve an efficient long-term level of conservation, restoration and sustainable use of biodiversity. Past efforts to address dryland forest loss have achieved much less than expected. Thus new paradigms are needed to go beyond the *status quo* with imagination and courage. Restoration and sustainable use of biodiversity are crucial to human well-being and poverty alleviation in Latin American dryland areas.

Forest restoration is frequently seen as a competing land-use type. Actually it competes primarily with low-profit land uses such as deforestation, to induce poor quality pastures for extensive ranging cattle or the establishment of rain-fed staple crops. A public policy aimed at fostering forest restoration could provide incentives for the intensification of agricultural and cattle ranching in areas where these activities may be most profitable, allowing passive and active restoration processes in vulnerable marginal and steep areas possessing shallow and poor soils. Incentives could be provided for development of more nature-friendly land-use systems, wherever they are economically viable, if they do not introduce market dependence, and if they help to release pressure on marginal areas (e.g. organic farming, organic ranching, holistic ranching, shade-grown coffee and other alternative productive systems. However, see Tejeda-Cruz *et al.* (2010), for a critical assessment of the purported association between shade-grown coffee plantations and biodiversity conservation).

Forest restoration may enhance provision of a number of ecosystem services that could be exploited by local landowners, either by direct *in situ* commercialization or through the mechanism of payment for ecosystem services (**Box 10.8**). Among the former, increases of forest cover may provide support for sustainable ecoturism developments (e.g. bird and wildlife watching, fishing, river rafting, scenic spots, rural livelihoods, archaeological and historical tourism). In Mexico the development of an effective scheme of payment for environmental services for landholders of small forest stands and restoration plots is still needed; currently it is applicable only to owners of large land properties, who are almost non-existent in the regions of study.

The use of firewood in rural areas is widespread and it can be expected that, as in other heavily rural and poor regions of the world, it will continue to be used in many households of the study areas for at least the next 20–30 years (Bailis *et al.*, 2005). Most rural communities still rely on the increasingly difficult-to-obtain and expensive supply of firewood to meet their energy needs in households, bakeries and pottery shops. Yet production of firewood or its more efficient use does not receive commensurate incentives.

On best practices for forest restoration and monitoring

A number of different practices are currently understood by both lay people and specialists when forest restoration is mentioned. Yet it is debatable whether they are sustainable or not. For example, consider the risks associated with the establishment of monospecific plantations of fast-growing pines or exotic trees. Native pines are highly valued in Mexico for timber production. Yet, owing to their long-term impacts on local and regional biodiversity, as observed in Chile and Mexico, monospecific plantations of pines and exotics should receive incentives only to be established in poor marginal areas where no conflicts with conservation or restoration using native species are envisaged.

Participation of landholders in the planning stages should result in their commitment to establish local nurseries for the propagation of native tree and shrub species that are chosen collectively. Owing to their rural livelihoods, landholders may have an interest in identifying native trees and shrubs that provide a variety of useful products, are well adapted to local conditions, are more easily propagated, and are already highly valued and managed by local communities (see Chapters 5–7, **Boxes 10.2**, **10.5** and **10.9**). As for the sites where restoration should be prioritized, which is a decision that should be made in conjunction with criteria and preferences of landholders and expert opinion (Wester *et al.*, 2003; Failing *et al.*, 2004, Orsi *et al.*, 2010; Chapter 9), it is suggested that river margins and high elevation deforested steep slopes should perhaps receive highest priority. Restoration strategies that enhance and exploit natural forest regeneration processes in fields and pastures should also be considered.

Box 10.9 The connection between university research and education/teaching and a rural community in Mexico: the case study of the Barrancas Environmental Restoration Research Station, Morelos, Mexico

E. Ceccon

Mexico stands out among Latin American countries as the one with the largest area subjected to land-use change between 2000 and 2005 (318, 533 ha; FAO, 2005b), mainly involving loss of tropical forest, where a large percentage of Mexico's biodiversity can be found. At the same time some 30,000 communities exist in the country that manage an area close to 100 million ha (Bray, 1995), which have always been marginalized from the global benefits of social development in Mexico.

With this perspective in mind an initiative was carried out by researchers at the National Autonomous University of Mexico (UNAM) and the inhabitants of the Cuentepec community in the state of Morelos to create the Barrancas Environmental Restoration Research Station of the Tembembe River (BERRS). The station was created in 2003 and its main principal objective was to restore an area of tropical dry forest that belongs to the Cuentepec community through research and education projects, and to implement sustainable and productive community participation projects. Cuentepec is one of the poorest and most marginalized communities in the state of Morelos, which suffers one of the highest levels of ecosystem degradation (Gómez-Garzón, unpublished). Through an agreement, the community offered by way of a loan (over a period of 30 years) to UNAM the use of localized lands on a pre-established and geographically-referenced map. The agreement was based on four main points:

1. The agrarian community of Cuentepec offered by way of a loan to UNAM 97 ha of land and the *ejido* offered 20 ha with land tenure security for 30 years.
2. The UNAM would establish four university programmes for the Cuentepec community: restoration, productive restoration, environmental health, and environmental education.
3. To establish a mechanism with interactive participation and community co-management based on five-year plans to make sure that the community yield or gain profits in return.
4. To establish a mechanism to resolve conflicts based on bipartite opinion (involving both the community and UNAM).

The objectives and goals of UNAM in the agreement are:

1. To offer a university environment for the conduct of research and training on ecological and productive restoration. The BERRS is to be directly connected to the MSc programme in Ecological Restoration offered by UNAM.
2. To restore degraded areas and native biological communities of the river banks and slopes of the River Tembembe within the research station.
3. To establish a new paradigm in university social involvement that derives benefits for the indigenous Cuentepec community. UNAM should drive the sustainable development of the Cuentepec community so that at the end of the 30-year period the community can manage the terrestrial and aquatic resources within their gorges in a sustainable way, in particular, the research station area and a natural area established and protected by the community.

The following points are summarized from Ceccon *et al*. (in press).

Goals achieved:

There were a considerable number of advances in relation to the physical and biological knowledge of the area and the propagation and planting of several tree species in the study area. At the community level, domestic and public demonstrative cisterns were constructed to capture rain

Box 10.9 (cont.)

water. An environmental education programme was initiated in high schools in the Cuentepec community (Ceccon and Flores-Rojas, unpublished).

In the productive restoration projects, the consumption and market for firewood in Cuentepec was evaluated. The results allowed an understanding of the importance of biofuels in terms of rural energetics. Experimental plantations were started in agricultural lands using sustainable agroforestry and agroecological systems that include native species for firewood and organic fertilizers for food crop cultivation (Vázquez-Perales *et al*., unpublished).

Lessons learned:

Many things have changed over the six years of the research station's existence. Several programmes have been completed. However, the community's active participation still needs to be increased. Only a small portion of the community is aware of and participates in the projects that are implemented. A number of critical issues have been brought to attention within the productive restoration system established in the *ejido* lands; currently some deterioration in the internal organization of the community is observed, which has led to polarized views and makes it very difficult to perform the required tasks. At the same time, resistance is also observed to effect changes in paradigms in agricultural practices among people belonging to the *ejido* owing to the complexities involved in some systems such as the nonexistence of a well-established market to sell the new agricultural products.

Conclusion:

The BERRS has definitely contributed to increasing the status of leadership of UNAM in the state of Morelos in issues related to ecology and society, and has established stronger and more permanent bonds with the community at Cuentepec. However, a number of conflicts still remain and hinder efforts to fully develop all needed biological studies. Additional research still needs to be conducted regarding the perception of productive and environmental problems by the local population.

Figure 1 Experiments established at the BERRS. Photo: E. Flores-Ramirez

Box 10.9 (cont.)

Figure 2 A peasant working in a productive restoration experiment in his *milpa* field at Cuentepec. Photo: R. Vázquez-Perales

Figure 3 An indigenous woman of Cuentepec. Photo: E. Ceccon

Box 10.9 (cont.)

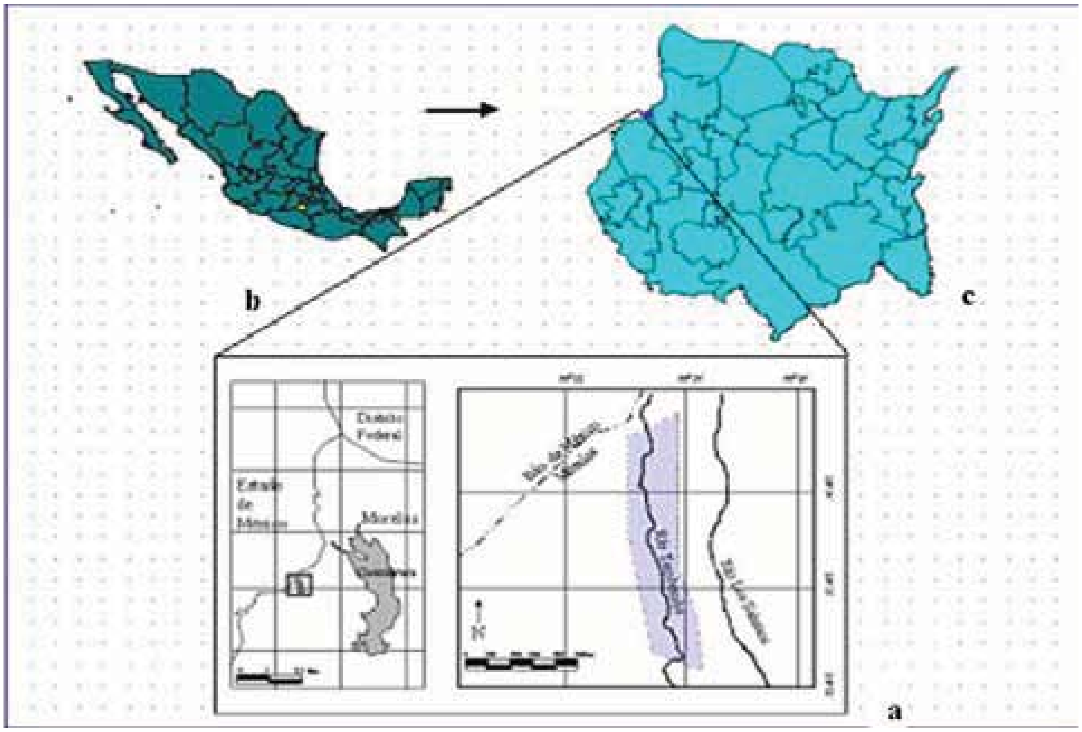


Figure 4 Map showing the location of the BERRS (after Galindo-Escamilla, unpublished).

Figure 5 Degraded land neighbouring the BERRS. Photo: E. Ceccon

Finally, a considerable number of restoration trials and programmes have been initiated in recent years in the regions of study, following a diversity of aims, methods, and results in their initial stages. This offers a richness of cases for potential meta-analysis (Rey Benayas *et al.*, 2009), whose value could be increased if restoration initiatives fulfil a minimum record-keeping from the onset and during their subsequent monitoring. The success of forest restoration initiatives will be dependent on effective monitoring in the field to examine the effectiveness of the management approaches adopted, and to define the mid- and long-term dynamics of landscape units as restoration takes place. If possible, it would be desirable to establish a protocol for a centralized, public access database on restoration plans that might be used in coming years to monitor restoration trials and predict the outcome of such initiatives. Supported by tools such as meta-analysis, this type of approach would greatly assist the development of a robust evidence base to support future decision making relating to restoration actions.

On education and research

A rich traditional knowledge is common to many communities within the study regions, particularly among those groups where indigenous ethnic and cultural elements have been maintained. It is currently considered that this tradition may provide a useful platform to improve human well-being and enhance biodiversity conservation. However, the maintenance of this tradition, which has among its major assets the use of high biodiversity, rests heavily on the elder members of the rural communities (**Box 10.10**). Market exposure or economic incentives may produce mixed effects on these communities (Godoy *et al.*, 2005). For example, new forestry practices may be fostered by external forces and these may be learned and valued by the new generations while ancestral knowledge fades into oblivion.

Box 10.10 Guidelines for restoration of native species in Mapuche communities

M.E. Gobbi

The Mapuche people (*mapu*: land, *che*: people), like all native peoples, have a long tradition of close relationships with nature. For them, plants are a source of medicine, food, as well as indicators of environmental conditions and/or of climatic changes. The Mapuches do not have a written language and, as a consequence, they have transmitted their knowledge about plants orally from generation to generation. Several reasons have caused the oral traditions to get lost or to be known only by the elderly: (i) expulsion from territories they originally inhabited, and then (ii) settlement in territories with different environmental characteristics, (iii) overexploitation of natural resources, owing to the lack of land for the development of communal productive activities, (iv) young people's migration to cities, (v) separation of families, etc.

My aim was to identify the main environmental problems, their principal causes, and possible actions to recover the degraded areas in the Currumil and Puel Mapuche communities, in northwest Patagonia. I studied the communities' perceptions of their own environmental problems, and encouraged knowledge circulation about native plants among the community members, and between the Mapuche community and the Universidad Nacional del Comahue committee. The particular aims were defined taking into account the specific needs of the Mapuche community members.

The work was sequentially organized: (1) meetings with the Currumil and Puel authorities: the *loncos* (Mapuche chiefs) and the community councils, (2) a workshop aimed to introduce and integrate the members of both communities, (3) specific workshops designed to satisfy the demands of each of the communities, and (4) coordination and support of the activities suggested

Box 10.10 (cont.)

in the workshops. The workshop topics were: environmental interpretation, main environmental impacts on nature, and the cultivation of native plants (**Fig. 1**). I designed a booklet containing basic information about each of these topics.

The activities performed were: (i) formulation of surveys designed and analyzed by members of the university committee and carried out by members of the Mapuche community, (ii) field trips intended to interpret nature in areas of the Mapuche community under some kind of productive use (cattle raising, camping sites, lagoon coasts, and a small tourist centre), and (iii) the writing of a book about the most common species and their uses within the Puel and Currumil communities.

The main results indicate that:

- Both communities are fully conscious of environmental damage and are concerned about its causes and consequences. They are also willing to apply management strategies that will allow them to recover these sites. They perceive that the present needs for land exploitation do not allow them to properly preserve these sites.
- Major knowledge of native flora is associated with the elderly in the community, both men and women.
- Among young people, there is some interest in this type of knowledge but they lack the mechanisms for transmitting it.
- Family orchards and greenhouses are quite frequent (57% of the families have greenhouses and 67% have orchards) and they are used exclusively for the production of exotic plants, mainly with food value.
- They do not cultivate native species or grow them on the lands surrounding their houses. Less than 10% of the people have had some experience in trying to propagate native plants. Most people (58%) expressed some interest in having native trees or shrubs in the proximities of their homes, and 44% were interested in learning about techniques to propagate native plants. In this sense, some ancient practices have been rescued and shared (**Fig. 2**) and some other ones, particularly associated with including some native species in their home yards, have been suggested.
- Community members selected 33 species for the elaboration of the material to be published in a book. The university committee took photos and wrote simple botanic descriptions and a glossary for these species; the members of the communities described the plant uses.
- The resulting book will belong exclusively to the communities and they will use it for educational purposes, and for exchanging information with other Mapuche communities. It will only be sold if they decide to do so.
- These plants are mainly used for medicine; secondly, they are also used for food and some species have religious or ceremonial value. Medicinal knowledge was generously shared with the university committee, but both Mapuche communities expressed strong resistance to making it public. In fact, in the book, we just mention 'medicinal uses' without giving any details concerning the elaboration of medicines or the kind of ailment they treat.

In conclusion:

i) In Puel and Currumil native plants are collected in natural conditions and only exotic plants with some medicinal or food value are cultivated.

ii) They find the idea of growing native plants both useful and new.

iii) During the course of the project, various other studies at different scales were proposed.

iv) Analysis and discussion of the main environmental problems, their causes and ways to solve them, allowed sharing of experiences and points of view, with the ultimate aim of finding alternatives to the use of natural resources.

Box 10.10 (cont.)

Other organizations and participating institutions: the BID's *Programa de Desarrollo Cultural* granted funds for producing the book on native flora (Project: *Rescatando saberes sobre plantas nativas de las comunidades mapuches Puel y Currumil*) and the *Universidad Nacional del Comahue* collaborated with field trips and workshops (Project: *Estrategias de manejo e innovación tecnológica para la sustentabilidad ambiental en territorio de comunidades mapuche*).

Figure 1 Workshop activities: (a) environmental interpretation, and (b) propagation of native plants; a Mapuche child bearing a juvenile of *Araucaria araucana*. Photo: a) A. Denegri; b) M. Gobbi

Figure 2 Repair of damage inflicted by goats (species introduced for wool and meat production) on *Araucaria araucana* trees, which is a tree highly appreciated by the Mapuche, both for religious and food values. They used traditional knowledge reported by the elderly in the community. Photo: M. Gobbi

Some of the uncertainty regarding potential outcomes can be attributed to methodological pitfalls or limitations of conventional research approaches to dealing with indigenous groups. However, it is also true that under current globalization pressures a wider cultural understanding about the relevance of trees and forest cover for economic, cultural, and recreational values is badly needed in both rural and urban environments. A community-wide awareness that forest restoration can be adopted as a commonplace activity that may generate permanent wealth and well-being needs to be enhanced. It is also important to promote the dissemination of knowledge on how to optimize local uses of forest products, on the scope for natural regeneration in passive restoration projects, limits to productivity and sustainable harvest, and the propagation of tree species in nurseries for restoration trials (Chapter 5, **Boxes 10.2**, **10.5**, **10.9**, and **10.10**). However, education and training should also reach technicians of the government agencies through *in situ* workshops and continuing education programmes on forest restoration and sustainable forest management.

The application of forest management plans and restoration practices often faces a lack of understanding of crucial ecological issues (**Box 10.6**). Although local interest and knowledge may exist on non-conventional uses of forest timber and non-timber products, more research is needed on their useful attributes, their current and potential economic value, and possible approaches for sustainable harvesting (Newton, 2008). Lessons learned from previous collaborative research on sustainable forest management highlight the value of integrated modelling tools for providing a framework to support the development and implementation of policy relating to sustainable forest management, including restoration actions (Newton *et al.*, 2007; 2009; Orsi *et al.*, 2010).

Conclusions

This chapter integrates research conducted by all ReForLan project partners with the aim of developing public policy recommendations, decision-support tools, practical guidelines and management plans for the restoration of dryland forest landscapes. A comparative approach is followed to show major differences, coincidences and potential impacts relating to these issues among the forest ecosystems in the six study regions where the project was conducted: central Veracruz (Mexico), the Upper Mixteca of Oaxaca (Mexico), the Central Depression of Chiapas (Mexico), the coastal range and Central Valley of Chile, north-west Argentina, and southern Argentina. The development and application of public policies and decision-support tools shows considerable variation among the study areas, even if they are located within the same country, as a result of major differences in social and economic development and native cultures at both national and regional levels. Yet the contrast among study regions provides an opportunity to identify major problems and recommendations to foster long-term restoration of dry forest landscapes, both in these areas and in others facing similar problems. Several case studies are presented to illustrate these issues, including two dealing with areas not covered in the main project (the state of Morelos in south central Mexico and Andean Bolivia). The focus of the chapter is on public policies, but other relevant topics such as outreach activities, education, elaboration of local guidelines and management plans are also discussed as examples of practical actions undertaken to support the implementation of forest landscape restoration concepts.

In conclusion:

- Public policies on forest restoration must continue to evolve in their aims, definitions and implementation procedures.

- Public policies on forest restoration must be agreed upon by all stakeholders. Eventually, grassroots groups and landholders should be given the opportunity to participate in a more active way, not only at the stage of initial consultation, but also during the decision-making and implementation processes.
- Rural livelihoods should be considered throughout the implementation of restoration programmes as a reference and source of novel alternatives for living in the rural environment in the future. However, the limited sample size that is typical in detailed socio-economic analysis should be borne in mind when extrapolating findings to a larger region or a wider context. Comparison of different study areas indicates that they are highly individualistic, and therefore restoration approaches will need to be tailored to the specific socio-ecological characteristics of each landscape.
- Academic groups must continue to develop and apply in pilot projects powerful state-of-the-art analytical tools aimed at disentangling the complexity of the issues involved in the long-term commitment of a forest landscape restoration programme.
- Academic groups should devote more effort to producing or supervising the elaboration and dissemination of educational materials with the potential to address a wide public, particularly on practical issues of forest management, restoration and ecosystem services of interest to landowners and technicians.
- Forest restoration programmes are local or regional social constructions involving all dimensions of a community experience: ecological, social, economical, cultural and political.

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11 SYNTHESIS: PRINCIPLES AND PRACTICE OF FOREST LANDSCAPE RESTORATION

A.C. Newton

Introduction

Recent years have witnessed a growth in interest in the science and practice of ecological restoration, in response to the environmental degradation that has occurred in many parts of the world. Restoration ecology is now firmly established as a scientific discipline (Aronson *et al.,* 2007; Clewell and Aronson, 2007), with publications regularly featuring in leading scientific journals (Dobson *et al.,* 1997; Lamb *et al.*, 2005; Roberts *et al.,* 2009; Rey Benayas *et al.,* 2009). Ecological restoration is now incorporated in a number of international policy initiatives, such as the Convention on Biological Diversity. This indicates that the potential contribution that restoration actions can make to both biodiversity conservation and improvement of human livelihoods is increasingly being recognized by policy makers (Nellemann and Corcoran, 2010; Secretariat of the Convention on Biological Diversity, 2010). At the same time, many hundreds of restoration initiatives have been implemented worldwide, representing an investment of billions of US dollars (Goldstein *et al.,* 2008).

Forest landscape restoration (FLR) can therefore be considered as one of many approaches to ecological restoration that are currently being implemented in different parts of the world. If FLR is to be adopted more widely, or even to be seen as the preferred approach to restoring forest ecosystems, then its effectiveness will need to be demonstrated. The principal aim of the research described in this book was to critically examine the principles underpinning FLR and how these may be applied in practice, and to provide a body of evidence regarding the effectiveness of the FLR approach. Specifically, the research explored application of FLR to dryland forests in Latin America, a forest type that is recognized as a global priority for biodiversity conservation, and of high importance for supporting human livelihoods (Miles *et al.,* 2006; Bullock *et al.*, 1995). Despite their importance, relatively little information is available regarding the ecological characteristics and dynamics of dry forests, particularly in the context of understanding the impacts of human activities and the potential for ecological restoration.

This chapter aims to integrate the information presented in preceding parts of the book, and to identify some of the key lessons learned. First, consideration is given to whether any general principles of FLR can be identified, which might be applicable to other locations where the approach could be implemented. Second, implications of the research for the practical implementation of FLR are examined. Third, some of the issues that emerged during the research are briefly considered, together with remaining knowledge gaps and critical areas of uncertainty. Some additional recommendations for policy and practice are then presented.

Principles of Forest Landscape Restoration (FLR)

The FLR approach has previously been described in detail by Lamb and Gilmour (2003), Mansourian *et al.* (2005) and Rietbergen-McCracken *et al.* (2007). No attempt is made here to summarize all of the information presented in these texts. However, these publications provide an appropriate foundation for identifying any common principles of FLR.

Maginnis and Jackson (2007) present four key features of FLR:

1. FLR is a process, which embodies three key principles: (i) it is participatory, (ii) it is based on adaptive management and is therefore responsive to social, economic and environmental change, and (iii) it requires a clear and consistent evaluation and learning framework.
2. FLR seeks to restore ecological integrity; simply replacing one or two attributes of forest functionality across and entire landscape tends to be inequitable and unsustainable.
3. FLR seeks to enhance human well-being, based on the principle that the joint objectives of enhanced ecological integrity and human well-being cannot be traded off against each other at a landscape scale.
4. FLR implementation is at a landscape scale; in other words, site-level decisions need to be made within a landscape context.

Some of these points require further clarification. First, adaptive management can be defined as the integration of design, management and monitoring to systematically test assumptions in order to adapt and learn (Newton, 2007). The approach involves systematically trying different management actions to achieve a desired outcome, and on monitoring results achieved to assess how they compare to those predicted at the outset. Adaptation involves changing assumptions and interventions in response to the information obtained as a result of monitoring. A monitoring programme is therefore essential if an adaptive management approach is to be effective, together with an appropriate evaluation and learning framework to ensure that lessons are learned from management experience.

The term 'ecological integrity' is repeatedly referred to in the FLR literature. Mansourian (2005) defines this as "*maintaining the diversity and quality of ecosystems, and enhancing their capacity to adapt to change and provide for the needs of future generations*". Lamb and Gilmour (2003) further expand on this definition, suggesting that it includes "*ecological authenticity (e.g. ecological naturalness, viability, health) as well as the functional effectiveness of the restoration process (e.g. the extent to which key ecological processes are regained)*". As pointed out by Newton (2007), terms such as 'authenticity', 'naturalness' and 'health' are poorly defined and are consequently difficult to measure; the same may therefore be said of 'ecological integrity'. If an adaptive management approach is required, then 'integrity' will need to be monitored: how can this be achieved in practice? Although Gasana (2007) provides some suggestions regarding potential indicators that could be used, the list of potential measures of 'integrity' is very long, and little information is available regarding the effectiveness or reliability of such measures as indicators of 'integrity'. If the key objective is to maintain the diversity and adaptive capacity of a forest ecosystem, then this will depend on the ecological processes influencing these properties. These could potentially include dispersal, colonization, regeneration, growth, competition and succession, among many others. Again, indicators could potentially be required for each of these, although some of these processes are very difficult to measure in practice (Newton, 2007).

Local nursery for propagating native tree species used for firewood plantations in Ocuilapa, Chiapas, Mexico. Photo: N. Ramirez-Marcial

The tropical dry forest study site identification. Photo: G. Williams-Linera

The third feature listed by Maginnis and Jackson (2007) focuses on enhancing human well-being. The linkage between human well-being and the condition of ecosystems is currently a major focus of research, as illustrated by the Millennium Ecosystem Assessment (2005a). Central to this research approach is the concept of 'ecosystem services', or the benefits provided by ecosystems to humans. FLR should therefore increase the provision of ecosystem services, by restoring those ecological processes and functions on which this provision depends (Fisher *et al.*, 2008). Maginnis and Jackson (2007) also suggest 'that the joint objectives of enhanced ecological integrity and human well-being cannot be traded off against each other at a landscape scale'. This depends on an implicit assumption that human well-being and ecological integrity are coincident within a landscape, an assumption that is largely untested. However, it is not difficult to envisage how conflicts could arise: human well-being is heavily dependent on access to food, which is generally more readily obtained from cropland than from forest. Evidence suggests that 'win-win' solutions between human well-being and ecosystem condition may be difficult to achieve in practice; tradeoffs may also need to be made between one ecosystem service and another (Tallis *et al.*, 2008; see also **Box 11.1**).

The problems associated with the features listed by Maginnis and Jackson (2007) suggest that there is a need to refine them further. Ideally, a set of core principles might be defined for FLR initiatives, which might provide guidance to application of the approach in any given situation. However, the most consistent result of the research presented in the preceding chapters is the overriding importance of local context. Each comparative analysis highlighted pronounced differences between case studies in terms of the factors responsible for forest loss and degradation; patterns of species richness and composition; use of forest resources by local communities; effectiveness of different restoration approaches; pattern and rate of forest recovery; policy context and recommendations; and the local value of different ecosystem services (**Box 11.1**). This important finding highlights the difficulty of developing a generally applicable procedure for implementing FLR; rather, what is a required is an approach that is sufficiently flexible that it can be adapted to any local situation. The main lesson learned from the current research is that FLR will need to be adapted to the particular circumstances of any given context, in order for it to be effective.

On this basis, I propose a revised set of principles for FLR, as follows:

1. FLR is a flexible process that will need to be adapted to each individual ecological, socioeconomic, cultural and political context.

2. FLR is a participatory process, requiring the engagement of stakeholders to be successful.

3. FLR should be based on an adaptive management approach to ensure that it is responsive to social, economic and environmental change; it therefore requires both an adequate monitoring programme and an appropriate learning process.

4. FLR seeks to restore ecological processes at the landscape scale that will ensure maintenance of biodiversity and ecosystem functions, and confer resilience to environmental change; this will require site-level decisions to be made within a landscape context.

5. FLR seeks to enhance the provision of ecosystem services to humans at the landscape scale, and thereby contribute to improved human well-being.

Lamb and Gilmour (2003) present a number of potential benefits of FLR, but fail to consider the relationships between these different elements. Here is a schematic diagram to illustrate these relationships (**Fig. 11.1**). FLR can potentially lead to a number of different impacts, including increased forest connectivity, increased provision of ecosystem services, mitigation of threats, etc. These may have additional impacts, for example on biodiversity and both economic and environmental risk. Together, such impacts may lead to policy-relevant outcomes, such as the alleviation of poverty, the development of sustainable livelihoods and conservation of biodiversity. This conceptual model offers a generally applicable description of the FLR process, and could potentially provide a framework for monitoring its impacts and effectiveness.

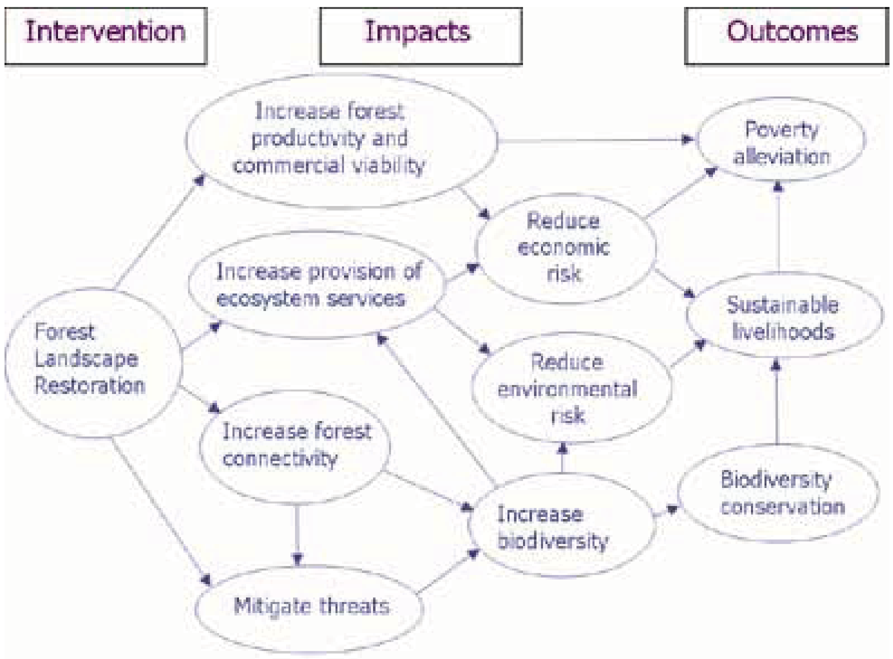


Figure 11.1 Schematic diagram highlighting the relationships between potential impacts of the Forest Landscape Restoration approach, and potential outcomes.

Box 11.1 Mapping and valuation of ecosystem services in dryland forest landscapes

J. Birch and A.C. Newton

Ecosystem services are the benefits that people obtain from ecosystems (Millennium Ecosystem Assessment, 2005). Dryland forests provide a number of important ecosystem services such as carbon sequestration, erosion control and biodiversity, which benefit local communities and other sectors of society. However, dry forests are highly threatened (Janzen, 1988) owing to unsustainable land-use practices, which include overharvesting of fuelwood and expansion of rangeland for livestock.

To prevent the loss of ecosystem services provided by dryland forests there is a need to understand how and where these services are created, and their value to human society. By assigning such services a monetary value, it is possible to communicate their importance to policy makers. To date, very few studies have attempted to value and map ecosystem services across a landscape, particularly in the context of forest restoration. While rapid progress has recently been made in understanding how ecosystems provide services and how such provision relates to economic value, it has proved more difficult to produce credible, quantitative estimates

Box 11.1 (cont.)

of ecosystem service values (Nelson *et al.,* 2009). The development of methods to provide such values is an area of active research, as illustrated by the work of The Economics of Ecosystems and Biodiversity (TEEB) initiative (Balmford *et al.,* 2008), and the Natural Capital Project (Daily *et al.,* 2009).

A key question is*: what impact does a particular policy or management intervention have on the provision of ecosystem services?* This requires analysis of the production of ecosystem services, the benefits provided to different stakeholders, and the costs that might be incurred. This can be achieved by comparing scenarios where a particular policy action has either been implemented or not (**Fig. 1**). Cost-benefit analysis of marginal changes will then enable any net benefits to be identified that might arise from implementing the scenario.

Using this method, a comparative study was undertaken across four of the ReForLan study areas to examine whether forest landscape restoration was cost-effective. Using ArcGIS, two land-cover maps were created, representing the current land-cover and a projected land-cover created using a spatially explicit model of forest dynamics, LANDIS-II (Mladenoff, 2004; see Chapter 8). The value per unit area of five ecosystem services (carbon, livestock, NTFPs, timber, tourism) was calculated for each land-cover type. The marginal value was calculated by referring to the change in area of each land-cover type between the two situations. This marginal flow of ecosystem services was subjected to various discount rates to allow for uncertainty in the application of discounting to environmental economics (Newell and Pizer, 2003; Rees *et al.,* 2007). The direct costs of implementing a restoration project and the associated opportunity costs (in this case, loss of livestock production) were calculated and a cost-benefit analysis method was used to estimate the marginal net present value (mNPV) of restoration. Geographic Information System (GIS) software was used to map mNPV on to the scenario land-cover map. By isolating the reforested areas where benefits outweigh the costs, policy makers can see where (and even if) forest restoration is economically viable for a particular landscape (see **Fig. 2**).

Our results showed that in all study areas, passive restoration was cost-effective. In contrast, active restoration proved costly and these costs frequently outweighed the benefits, especially at positive discount rates. Our results demonstrate that forest restoration is highly dependent on the local context, both in terms of the relative costs of restoration actions, and the potential value of different benefits. Since financial resources for conservation action are limited (Margules and Pressey, 2000), careful targeting of restoration actions will therefore be necessary to ensure that the approach is cost effective.

Full details of this research are presented by Birch *et al*. (2010).

Box 11.1 (cont.)

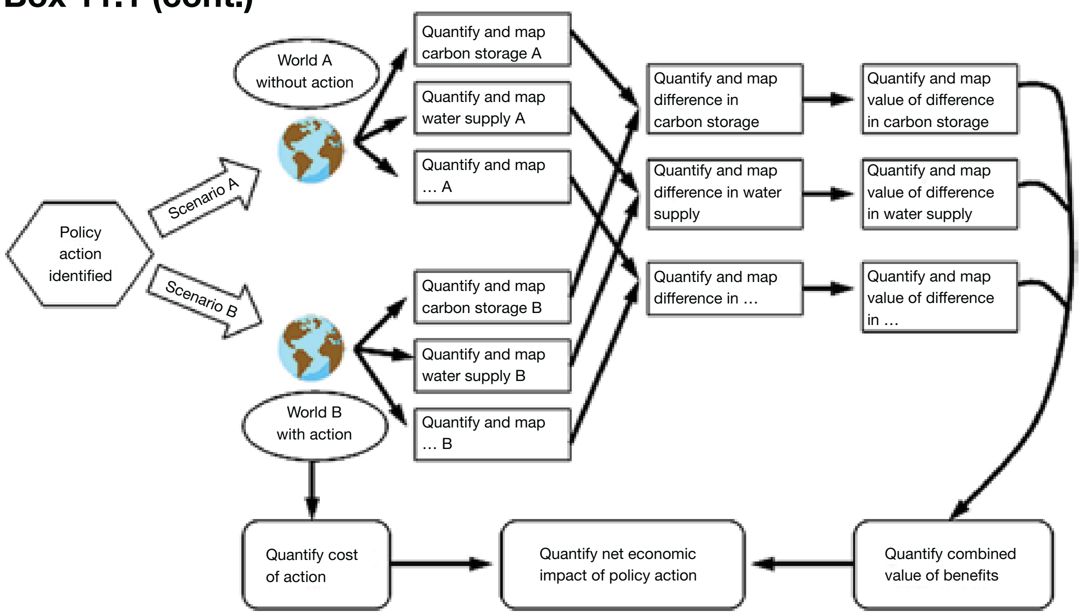


Figure 1 Analytical framework developed by Balmford *et al.* (2008) for the spatial analysis of ecosystem services, providing a practical tool for evaluating the cost-effectiveness of different policy interventions, at any scale. The framework focuses on quantifying the marginal costs and benefits associated with changes in ecosystem services. Marginality is essential because at all times the relevant question is what is the difference in benefits and costs from the implementation, or not, of a particular policy action. Quantifying marginal costs and benefits requires contrasting two counterfactual 'states of the world', defined in terms of variables such as land-cover, climate, human distribution and human activities. The two states should be identified through the development of scenarios (i.e. descriptions of plausible alternative futures), and must be carefully matched, such that they differ only in the implementation or not of a particular policy action. By coupling these contrasting sets of conditions with spatially-explicit models of service production and value, the overall economic consequences of any given policy intervention can then be estimated – a significant advance over most previous valuation approaches, which have largely been static rather than dynamic.

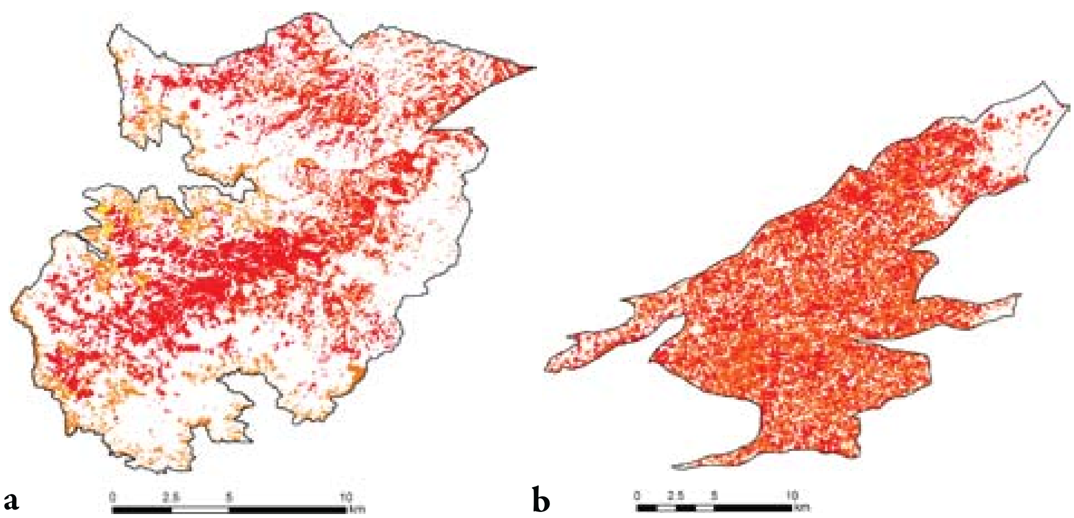
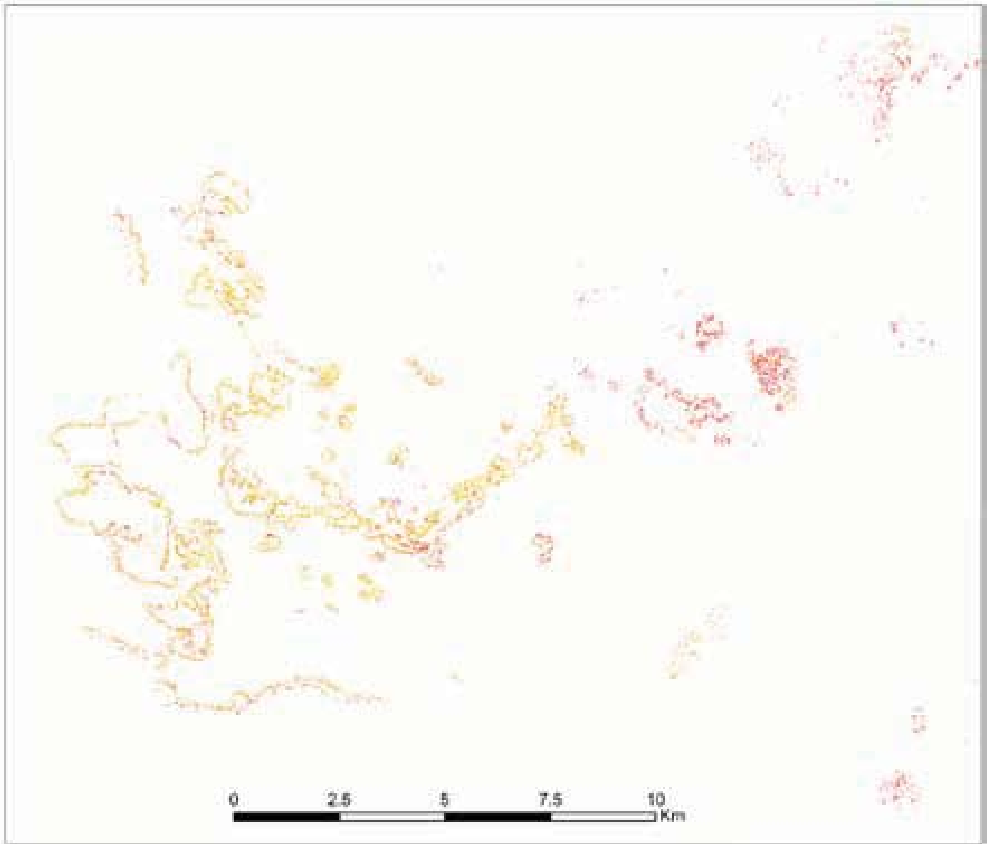


Figure 2 Maps illustrate the areas of net positive value under the restoration scenarios for each of the four study areas: (a) El Tablon, Mexico; (b) Central Veracruz, Mexico. Maps are represented for a 20-year time horizon at 5% discount rate. Higher values are illustrated in red and lower values in yellow. Maps (c) and (d) on page 360.

Box 11.1 (cont.)



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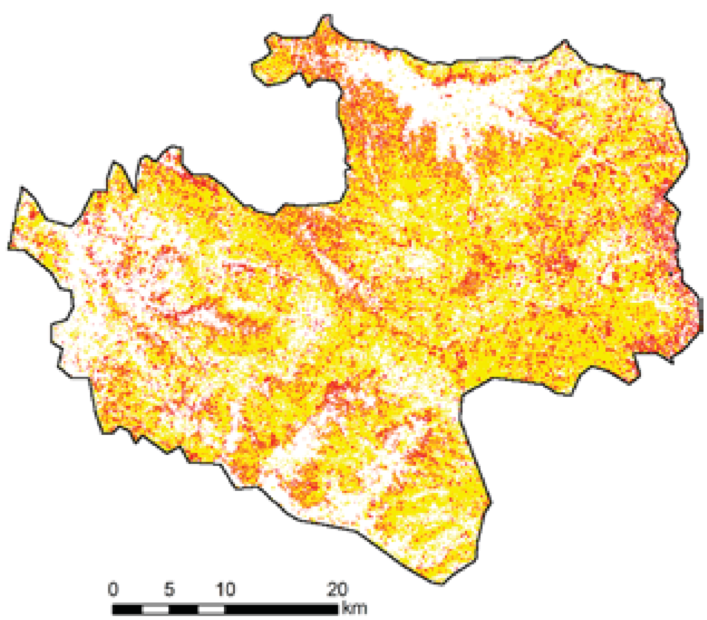

d

Figure 2 (cont). Maps illustrate the areas of net positive value under the restoration scenarios for each of the four study areas: (c) Nahuel Huapi, Argentina; (d) Quilpue, Chile. Maps are represented for a 20-year time horizon at 5% discount rate. Higher values are illustrated in red and lower values in yellow.

Vista Barranca de Acazonica. The study area is located in the tropical dry forest region in the municipalities of Paso de Ovejas and Comapa, Veracruz, Mexico. Photo: G. Williams-Linera

Bee-keeping in Colliguay valley, Chile. Photo: J. Birch

Achieving FLR in practice

Each of the preceding chapters identified a number of findings and recommendations with implications for the practical implementation of the FLR approach (see Chapters 2–10). These may be summarized as follows:

- Understanding the factors responsible for forest loss and degradation is essential for FLR approaches to be developed (Dudley, 2007).This research demonstrated that remote sensing and GIS techniques can be used together with statistical modelling approaches to identify both the pattern and the proximate causes of forest loss. Results indicate that land-use intensification continues to threaten dryland forest cover in many regions of Latin America. Forest loss was generally found to be more likely on gentle slopes, but overall, the role of different factors responsible for forest loss varied markedly between study areas. Such analysis can inform the development of restoration strategies and plans, by identifying those threatening processes that need to be addressed if restoration actions are to be successful.
- Analysis of remote sensing imagery using GIS indicated that dryland forests have exhibited a progressive fragmentation and degradation in most, but not all, of the study areas examined. FLR approaches have the potential to address both the fragmentation and degradation of forest that were documented here in multiple study areas. Such interventions should be planned and implemented at the landscape scale, to ensure they are effective in increasing connectivity among forest patches.
- The factors affecting patterns of species composition and richness were found to vary markedly between the different study landscapes.The effect of site (country and province) was found to be stronger than other factors, highlighting the importance of the local context when identifying restoration priorities. Results suggest that restoration is likely to be more successful in terms of impact on species richness when restoration activities are conducted at higher elevations than in lowland areas, and when the size of the remnant fragments is relatively large. However, the conditions of each landscape must be analyzed separately. Similarly, the history of local anthropogenic disturbance should be investigated to fully understand the processes that account for the present patterns of species richness in each region. Such an understanding is necessary if FLR is to be effective in restoring biodiversity.
- The results of field experiments established in each study area led to the identification of some key ecological processes that limit the establishment and growth of threatened and/or socioeconomically important native tree species found in dry forests. Furthermore, these experiments enabled the identification of restoration techniques that can overcome some of those constraints. In all of the study areas the main limiting factor for establishment was found to be drought. Restoration efforts in drylands have to confront the long dry season affecting seedling survival during any transplanting effort.Transplantation should take place at the beginning of the rainy season, or be undertaken in rainy years. Supplemental irrigation may also be effective in supporting tree establishment.The use of nurse species was found to be important for protecting seedlings from desiccation, thus improving seedling survival and initial growth. Exclusion of large herbivores is also often essential for successful tree establishment.
- While native tree species were preferred in restoration trials, exotic species are recognized as important in some circumstances, such as in very degraded sites. Native tree species

displayed higher survival rates than exotic species, especially during the dry season. Results from most field experiments suggested that natural regeneration can be encouraged by protecting successional areas from herbivores, fire, and selective cutting; enrichment planting is an appropriate method in early successional sites lacking non re-sprouting and key primary tree species; and mixed species plantations can be established on highly degraded sites. Local knowledge must be taken into account for the selection of tree species; local people should participate in the selection process and be aware of the importance of forest recovery owing to the environmental services that it provides. However a lack of knowledge of the biology of native tree species and secondary successional processes limits their implementation in management and conservation plans. Information about native trees is restricted to a few species; it is therefore paramount to carry out plant species research that addresses their phenology, seed dispersal, germination, growth, and vegetative reproduction.

- Dryland forest species can be an important source of economic resources to local people and provide valuable ecosystem services. The value of such species can be identified through interviews and workshops with local people. Although many native tree species of dryland forest in the study areas were recognized as useful, knowledge about their use value is being lost and, at least in some cases, is unevenly distributed among local people. Furthermore, formal commercialization of such products is uncommon. Firewood is one of the most widespread uses in all the study areas, but actions aimed at replenishing the losses owing to extraction are either non-existent or insufficient. Factors responsible for forest loss and degradation include the loss of awareness of the importance of native forest species among local people, disengagement of formal education from local knowledge and traditions, insufficient information on the potential economic or ecological importance of native plants, lack of commercialization channels for native forest products, conflicting governmental policies, the introduction of exotic species, and a lack of coordination among stakeholders involved in forest management and conservation. These problems need to be addressed in future FLR initiatives.

- Forest fragmentation and degradation affect patterns of genetic variation at different scales. The strength of such impacts particularly depends on autoecological characteristics of the studied species, historical events that have shaped gene pools, landscape-use change, environmental gradients, and disturbance regimes that are key determinants of such shifts. FLR approaches should consider patterns of within- and between-population genetic variation which in turn will be key determinants of restoration success.

- Spatially explicit modelling tools can be used to explore the potential for restoration of forest landscapes. Specifically, the modelling approach used (LANDIS-II) enabled projections to be made regarding the pattern of regeneration and spread of native forest under different anthropogenic disturbance regimes, providing insights into the potential for passive restoration approaches. Modelling results indicate that dry forest can be resilient to some forms of anthropogenic disturbance, and that different forms of disturbance may have interactive effects on ecological patterns and processes. Examples demonstrate how spatial models can inform approaches to forest landscape restoration, by indicating those locations within a landscape where particular restoration approaches are most likely to be successful.

- One of the most important decisions with regard to FLR programmes is where restoration actions should be undertaken to obtain the best results. However the identification of forest restoration priorities is complex. First, the objectives of the restoration effort need to be clarified. Different objectives usually involve different priorities. A sound analysis of such objectives is therefore a necessary pre-condition to the success of any restoration plan. Once the objectives have been identified in consultation with stakeholders, decision analysis techniques may be applied to the actual definition of restoration sites. Multicriteria evaluation (MCE) techniques are particularly suitable for their ability to combine multiple decision criteria, incorporate the values of different stakeholders and deal with spatially-explicit information. However, these methods should be handled carefully to provide proper support to decision making. The final outcome should not be seen as the best solution, but rather as the most suitable option in the light of the value judgements expressed by the stakeholders. Finally, FLR is carried out for people and with people, implying that the voice of local communities and other stakeholders must be taken into account throughout the entire process.

- The development and application of public policies and decision-support tools shows considerable variation among the study areas, even if they are located within the same country, as a result of major differences in social and economic development and native cultures at both national and regional scales. Overall, public policies on forest restoration must continue to evolve in their aims, definitions and implementation procedures. Policies on FLR must be agreed upon by all stakeholders. Grassroots groups and landholders should be given the opportunity to participate in a more active way, not only at the stage of initial consultation, but also during the decision-making and implementation processes. Rural livelihoods should be considered throughout the implementation of restoration programmes. Comparison of different study areas indicates that they are highly individualistic, and therefore restoration approaches will need to be tailored to the specific socio-ecological characteristics of each landscape.

Emerging issues, knowledge gaps and critical uncertainties

This section examines some of the cross-cutting issues that emerged during the research, which have not been addressed in detail by the preceding chapters. Some of these were identified in discussions held during project workshops, on which the following account is based. Attention is also given here to those issues where knowledge is particularly lacking, and which are therefore associated with a high degree of uncertainty. These issues might usefully be the focus of future research.

1. Where to restore?

As noted earlier, a key decision regarding FLR programmes is prioritizing areas for restoration. For example, should areas that have been completely deforested be given highest priority, or should priority be given to those sites that are degraded, but where some forest remains? When discussed at a project workshop, little consensus emerged on this point. This is largely attributable to the issue of feasibility: while restoration of entirely deforested sites might be more desirable, it would typically be more difficult to achieve on such sites. When these two options were considered separately, the relative importance of different criteria for site selection was found to differ among the respondents involved in the workshop exercise (**Box 11.2**). In addition, there was relatively little agreement among this group of experts regarding different criteria for the specific case of restoration in deforested areas. Should sites that are located relatively close to existing forest be prioritized over those that are more distant? And should unforested sites that are heavily degraded be prioritized over those that are less degraded? For both of these questions, there was no consensus among respondents (**Box 11.2**). It is possible to argue, for example, that sites located close to existing forest are likely to be colonized more rapidly by species, increasing the chances of restoration being effective. On the other hand, restoration of isolated, highly degraded sites might provide greater net benefits, in terms of improving environmental condition, provision of ecosystem services, etc.

Box 11.2 Where should biodiversity be restored in the drylands of Latin America? Findings of the ReForLan expert workshop

D. Geneletti, F. Orsi

During the 3rd ReForLan Project Meeting (Trento, September 21–27 2009), a one-day workshop was held with the aim of gathering expert opinion to support the identification of forest restoration priorities in the drylands of Latin America. The workshop focused on one specific objective of forest restoration interventions (biodiversity conservation), one specific region of the world (Latin America), and one specific ecosystem type (drylands). The purpose was to identify general principles that can be considered as applicable throughout the region, without making reference to local conditions in specific study areas.

The workshop consisted of the following stages:

1. Individual questionnaire distributed to workshop participants.
2. Presentation of overall questionnaire results, including agreement rates to the different questions.
3. Discussion of the most critical and less agreed-upon issues.
4. Revision of questionnaire answers and final results.

The questionnaire built on the results of the expert survey conducted during the project (see Chapter 9), through which a number of criteria to be considered when selecting restoration priorities were identified. The criteria identified during the survey were:

a. Connectivity

b. Degradation

c. Disturbance

d. Diversity (at the ecosystem level)

e. Diversity (at the species level)

The questionnaire aimed to take this process a step further, by collecting opinions on how to actually assess such criteria and on their relative importance in selecting restoration priorities. Each criterion was assessed with reference to two different approaches to forest restoration interventions: restoration of unforested areas and restoration of degraded forest areas.

The workshop findings are summarized in **Figs. 1** and **2**. **Fig. 1** presents an overview of the interpretation of the five criteria under the two restoration approaches, together with the agreement rate that characterized the assessment of each criterion. As can be seen, according to the experts, during the restoration of unforested areas priority should be given to sites that are close to existing forest, located in less degraded areas that are less exposed to ongoing pressures, and characterized by high diversity in terms of both species and ecosystems. Among the five criteria, connectivity and degradation are those with the lowest agreement rate (around 60–65%). The identification of restoration priorities in degraded forest areas should follow the

Box 11.2 (cont.)

same principles except for the fact that more degraded areas should receive priority. In general, the agreement rate that characterized criteria assessment for degraded forest areas was higher than in the previous case, and always above 70%.

	Restoration of unforested areas	Restoration of degraded forest areas
Isolated or close to existing forest?	Close to existing forest	Close to existing forest
In areas more or less degraded?	In less degraded areas	In more degraded areas
In areas more or less exposed to on-going pressure?	In less exposed areas	In less exposed areas
In areas with high or low ecosystem diversity?	In areas with high diversity	In areas with high diversity
In areas with high or low species diversity?	In areas with high species diversity	In areas with high species diversity

Agreement rate
>80%
65-80%
50-65%

Figure 1 Results of the criteria assessment and agreement rate.

Fig. 2 presents a qualitative overview of the weight of the different criteria. In both restoration conditions, degradation and disturbance are considered to be very important factors to guide the selection of restoration priorities. Connectivity is considered to be very important in the case of restoration in unforested areas, but slightly less important for degraded forest areas. Ecosystem diversity is scarcely important for either restoration approach. Finally, species diversity is considered very important to guide the selection of restoration priorities in degraded forests, but not so important in unforested areas.

	Restoration of unforested areas	Restoration of degraded forest areas
Connectivity	+++	++
Degradation	+++	+++
Disturbance	+++	+++
Ecosystem diversity	+	+
Species diversity	+	+++

+++ Very important
++ Important
+ Not so important

Figure 2 Results of the weight assessment.

Box 11.2 (cont.)

Table 1 Number of times (%) in which each criterion ranked in each position (Restoration of unforested areas).

	1st	2nd	3rd	4th	5th
Connectivity	53	7	7	20	13
Degradation	36	36	7	14	7
Disturbance	33	7	40	20	0
Ecosystem diversity	20	0	27	27	27
Species diversity	13	13	6	31	38

Table 2 Number of times (%) in which each criterion ranked in each position (Restoration of degraded forest areas).

	1st	2nd	3rd	4th	5th
Connectivity	38	19	6	13	25
Degradation	29	18	12	18	24
Disturbance	27	13	33	27	0
Ecosystem diversity	21	7	36	29	7
Species diversity	8	31	15	15	31

These results highlight the difficulty of obtaining consensus even among a single group of stakeholders (restoration scientists) when faced with decisions about the purely technical aspects of ecological restoration. This issue was highlighted further by results of the Delphi survey conducted with the global community of restoration scientists (Chapter 9, Orsi *et al.*, 2010). Part of the problem is semantic: even words as commonly used as 'restoration', 'degradation' and 'biodiversity' are interpreted variously by different researchers. As noted by Peters (1991), the vague or imprecise definition of terms is a common problem in ecological science, making such concepts difficult to operationalize in practice. When the broader pool of stakeholders with an interest in FLR is considered, the problem of reaching a consensus is going to be even more difficult to achieve. This was illustrated by the results of the stakeholder surveys described in Chapter 9. The lack of a clear scientific consensus regarding which sites should be prioritized for restoration further emphasizes the importance of involving multiple stakeholders in setting such priorities. Nevertheless, if agreement can be reached on choice of criteria and indicators, then the implications of different values (or weights) among stakeholders for the prioritization process can potentially be explored through the use of mapping tools (**Box 11.3**).

Box 11.3 An integrated approach to identifying restoration priorities in dryland forest landscapes

E. Cantarello, F. Orsi, D. Geneletti, A.C. Newton

In Chapter 9, a series of criteria and indicators were developed, with the aim of identifying priorities for restoration of dry forest. This example illustrates the application of selected criteria and indicators, which are applied here in combination to provide an integrated analysis. In each case, individual GIS layers were produced for each indicator, which were then overlaid to provide a single map of restoration priorities for each study area (**Table 1**, **Figs. 1–4**).

This approach can potentially be used to help identify priority locations for restoration actions within a landscape. However, the results will differ depending on the choice of criteria and indicators, and on the relative weights applied to each (**Table 1**). Such decisions should be made as part of a participatory approach involving stakeholders. The maps provided here are therefore only illustrative; however they indicate the potential value of this approach as a tool to support decision making. Potentially, such maps could be combined with outputs of cost-benefit analyses (see **Box 11.1**) to identify high priority areas for restoration, where such restoration is likely to be cost effective.

Table 1 Details of prioritization exercise undertaken for each of four study landscapes using the software ILWIS Open v 3.6 (©52 North, March 2009, Germany). The following criteria, indicators and spatial layers were employed following Orsi *et al.* (2010).

Criteria	Indicators	Weight	Spatial layers
B.1 Connectivity-corridors	B.1.3 Proximity index	0.24	Proximity index as calculated in FRAGSTATS (v3.3) (McGarigal *et al.*, 2002), with a moving window of 300 m.
B.2 Degradation	B.2.1 Patch area	0.25	Patch area as calculated in FRAGSTATS for the forest patches larger than 0.45 ha.
B.3 Disturbance	B.3.3 Road density (distance to roads)	0.24	Euclidean distance to all roads as calculated in ArcGIS 9.2 (© 1999–2006 ESRI Inc. California, USA).
B.4 Diversity (ecosystem/ landscape level)	B.4.2 Elevation heterogeneity	0.065	Standard deviation of the digital elevation model as calculated by using the focal statistics tool of ArcGIS and neighbourhood settings of 3 x 3 cells in central Veracruz and Quilpue, 5 x5 in Tablon and 7 x7 in Nahuel Huapi.
	B.4.1 Aspect heterogeneity	0.065	Variety of the aspect raster map as calculated using the focal statistics tool of ArcGIS and neighbourhood settings of 3 x 3 cells in central Veracruz and Quilpue, 5 x5 in Tablon and 7 x7 in Nahuel Huapi.
B.5 Diversity (species level)	B.5.2 Tree species richness	0.15	Species richness. In central Veracruz, the species richness map was derived from the outputs of GARP models (Stockwell and Peters, 1999), which were produced using the software DesktopGarp (©2002 University of Kansas Center for Research, Inc., USA). Only the 22 most abundant species were considered. In Tablon and Quilpue, the species richness map was derived from the ecoregion map of LANDIS-II. Only the 25 most abundant species with establishment probability higher or equal to 0.7 were considered in Tablon, whereas all the species recorded in the field survey with establishment probability higher or equal to 0.7 were considered in Quilpue. In Nahuel Huapi, the establishment probability map was produced using the Mahalanobis distance of the only species present, and was used instead of a map of species richness.

Box 11.3 (cont.)

Figs. 1–4 Maps illustrating variation in the priority for forest restoration across four study landscapes. The maps present a combined index of forest restoration priority, produced using the weights and indicators listed in **Table 1**.

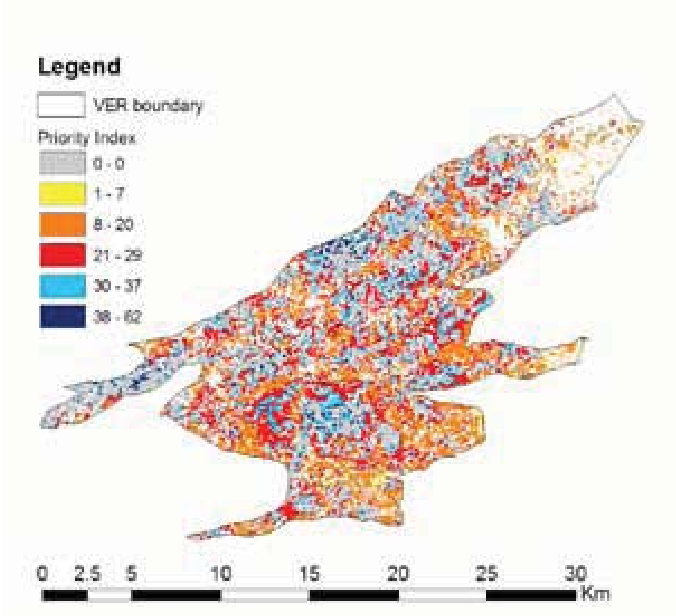

Figure 1 Priority map for the Central Veracruz study area. Priority index (1 to 100, with 100 the highest priority for restoration); grey areas are already forested; white areas represent non-active ecoregions and were excluded from the analysis.

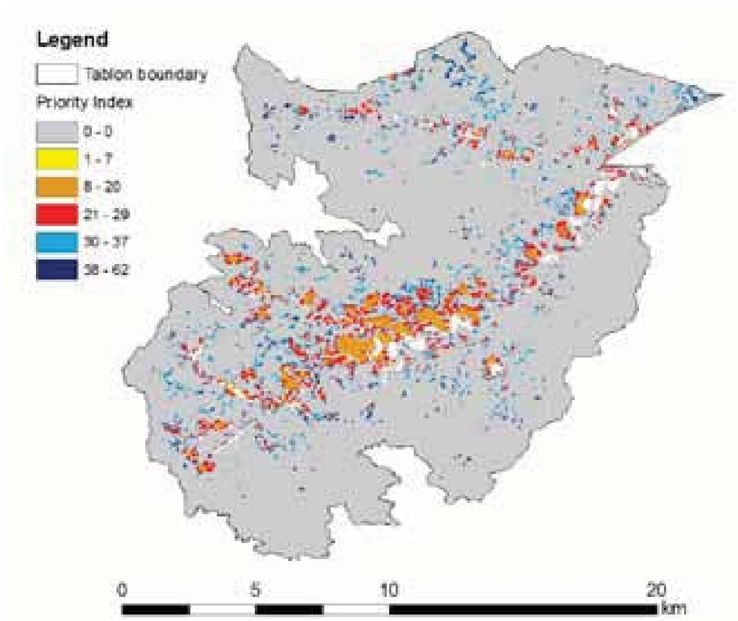

Figure 2 Priority map for the Tablon study area. Priority index (1 to 100, with 100 the highest priority for restoration); grey areas are already forested; white areas represent non-active ecoregions and were not included into the analysis.

Box 11.3 (cont.)

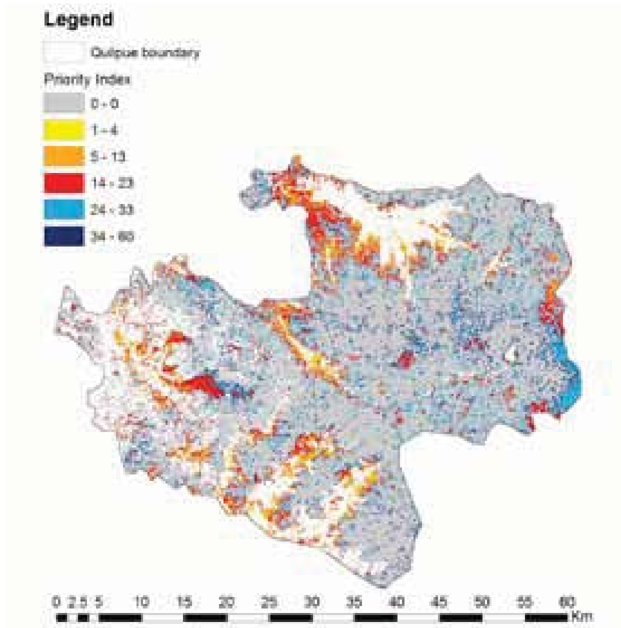

Figure 3 Priority map for the Quilpue study area. Priority index (1 to 100, with 100 the highest priority for restoration); grey areas are already forested; white areas represent non-active ecoregions and were excluded from the analysis.

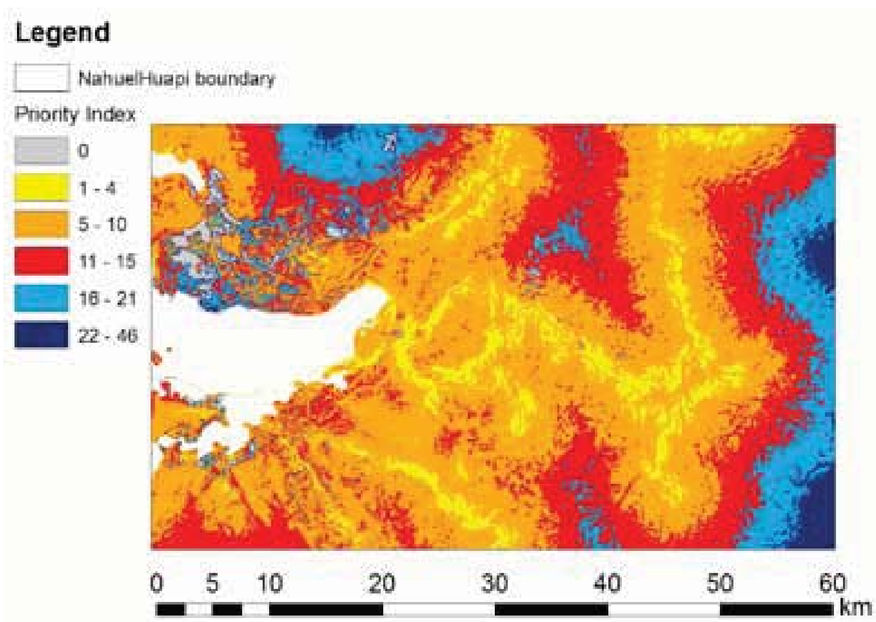

Figure 4 Priority map for the Nahuel Huapi study area. Priority index (1 to 100, with 100 the highest priority for restoration); grey areas are already forested; white areas represent non-active ecoregions and were excluded from the analysis.

2. Are there tipping points in degradation of forest ecosystems?

Tipping points occur when ecosystems shift to alternative states from which it may be difficult or impossible to recover. The identification of such tipping points is now an active area of research, and the focus of some policy concern (Secretariat of the Convention on Biological Diversity, 2010). Degradation may be particularly persistent in dryland ecosystems, leading in some cases to desertification (Millennium Ecosystem Assessment, 2005b). Such degradation can be intensified by positive feedbacks between organisms, soil and climate (Millennium Ecosystem Assessment, 2005b). This is not an issue that was explicitly examined by the research described here, but clearly has implications for restoration practice. In particular, is it possible that some sites are so degraded that ecological restoration would be impossible? Workshop discussions on this point concluded that this was generally believed not to be the case, at least for the case studies examined here. It was suggested that sites demonstrated the potential to recover at least to shrubland, even if very compacted and eroded (although sites subjected to salinization might be an exception, these were not explicitly examined in this research).

Another key question relates to whether restoration is intrinsically slower, or more difficult, on dryland sites, with either active or passive techniques. In general, little evidence was found in support of this suggestion; results from both experimental and modelling studies demonstrated the potential for successful restoration using both passive and active approaches. Preliminary comparison of survival and growth rates of restoration trials with those undertaken in humid forest types in the same regions (González-Espinosa *et al.,* 2007) do not indicate any major systematic differences, although this is an issue that could be investigated in greater depth in future. However, results do suggest that there may be a particularly high risk of failure on dryland sites, particularly using active restoration approaches (such as tree planting) on sites subject to severe drought.

3. Will FLR restore ecosystem function?

Another important issue not considered by the current research is whether FLR approaches will be effective in restoring functional ecosystems. This is key objective of FLR (Maginnis and Jackson, 2007), but one that is difficult to monitor. Furthermore, there is an issue of timescale: functional processes such as those relating to carbon, nutrient and water cycles are likely to require many decades to develop as forest stands mature. This issue is also directly relevant to the provision of ecosystem services, which is dependent on such functions (Fisher *et al.,* 2008). The analyses presented in **Box 11.1**, for example, are based on the assumption that restoration of forest cover, structure and composition will be associated with the restoration of ecosystem functions relating to ecosystem service provision, but this remains untested. The relationship between biodiversity and ecosystem function is another area of uncertainty; this again has been a major focus of research in recent decades, with conflicting results. For example, it might potentially be possible to restore some ecosystem functionality (such as carbon sequestration) without restoring native biodiversity, through the establishment of a stand of exotic tree species.

To address this knowledge gap, Rey Benayas *et al.* (2009) performed a meta-analysis of 89 restoration assessments undertaken worldwide in a wide range of ecosystem types, and found that restoration increased provision of biodiversity and ecosystem services by 44% and 25% respectively, based on an analysis of response ratios. However, values of both remained lower in restored than in relatively intact ecosystems. This investigation provides an important body of evidence indicating that ecological restoration is generally effective in restoring both ecosystem services and biodiversity. Furthermore, restoration actions focusing on the restoration of biodiversity are likely to lead to increased provision of ecosystem services. These findings support the suggestion that FLR approaches are likely to be effective in achieving these goals, but clearly this requires further examination. This could only be achieved by examining the progress of FLR interventions over a timescale of several decades; currently, given the relative novelty of the approach, case studies with which to conduct this analysis do not currently exist. Evidence could potentially be generated, however, by careful monitoring of any FLR initiatives that are undertaken, as part of their implementation.

4. Is FLR cost effective?

Even if FLR is effective in providing the intended benefits, will these be achieved at a reasonable cost? In other words, will the benefits outweigh the costs? One clear lesson from restoration actions undertaken in different parts of the world is that they are often expensive in financial terms. It is also important to note that there may be additional hidden costs. Any introduction of FLR will represent some form of change in land use, which will be associated with a cost to the landowner or land user. As an illustration, in all of the case studies explored here, expansion of forest area is likely to reduce the land available for livestock grazing or cultivation (**Box 11.1**), which will likely reduce income to local farmers. Such opportunity costs have been neglected by previous research on FLR (Mansourian *et al.,* 2005; Rietbergen-McCracken *et al.,* 2007), but as indicated here, could be substantial.

In fact, very few previous attempts have been made to perform a cost-benefit analysis of restoration projects. In a review of over 2000 restoration case studies, TEEB (2009) found that less than 5% provided meaningful cost data, and none provided analysis of both costs and achieved or projected benefits. The approaches for mapping and modelling multiple ecosystem services developed here (**Box 11.1**) provide a useful means of estimating such benefits, and when combined with estimates of costs, enable a cost-benefit analysis (CBA) of restoration actions to be performed. The results obtained suggest that passive restoration approaches are likely to be cost effective, but the higher costs of active restoration approaches may often outweigh the benefits. Another key finding is that cost effectiveness varies across a landscape; there may be locations within a given landscape where even active approaches may be cost effective (**Box 11.1;** Birch *et al.,* 2010). Further information is clearly needed on the relative cost effectiveness of different restoration methods in different contexts, and on their relative risks of failure.

Cost-benefit analysis of restoration initiatives is therefore in its infancy, and the analyses presented here must be viewed as preliminary. They are also based on a number of assumptions (Birch *et al.,* 2010), and are therefore associated with a degree of uncertainty. Key research needs for the future include analysis of the relationship between ecosystem service provision and ecosystem condition; the interactions and feedbacks between different services; and the potential need for tradeoffs between different ecosystem services, and between ecosystem services and biodiversity. The temporal dynamics of ecosystem flows are also poorly understood; some of the benefits of restoration could take many years to become apparent. In particular, the process of valuing ecosystem services is subject to a great deal of uncertainty, and little information is currently available regarding the distribution of beneficiaries of different services. Ensuring an equitable distribution of both the costs and benefits of restoration is likely to be one of the key challenges facing any future FLR initiative.

Dry forest in Colliguay valley, Chile. Photo: J. Birch

5. How can the costs of FLR be met?

Financial mechanisms are required to cover the cost of restoration actions, to compensate landholders for any loss of income, and to provide an incentive for landholders to engage in restoration actions. The problem of financing restoration initiatives has long been recognized, and a number of recent studies have explored potential options (Holl and Howarth, 2000; Milton *et al.,* 2003; Clewell and Aronson, 2007; Goldstein *et al.*, 2008; Janzen, 2002). Potential approaches include improved markets and payment schemes for ecosystem services (Jack *et al.,* 2008). Within the case studies examined here, little experience is available regarding the introduction or effectiveness of such schemes, at least in the context of dry forest restoration. Such schemes should ideally be based on a comprehensive analysis of the value of different ecosystem services (**Box 11.1**), although again, identifying an appropriate distribution of benefits will constitute a significant challenge.

One particularly promising mechanism is REDD+ (Reducing Emissions from Deforestation and Forest Degradation), which aims to offer incentives for developing countries to invest in low-carbon approaches to sustainable development (**Box 11.4**). Developed within the UN Framework Convention on Climate Change (UNFCCC), this provides a potential funding mechanism for supporting forest restoration activities, as a contribution towards the goal of enhancing forest carbon stocks. Revenues could be generated from the global market in carbon, which had reached US$ 125 billion by 2008; funding for REDD itself has already reached US$ 6 billion (Stickler *et al.,* 2009). REDD+ essentially offers an opportunity for stakeholders around the world to contribute financially to forest restoration. The mechanism has been criticized, however, for its focus on a single ecosystem service (carbon); there is a possibility that other services and social issues could be neglected or adversely affected (Stickler *et al.,* 2009). Potential negative social impacts include loss of livelihoods or access to lands undergoing restoration, a risk that is particularly high in areas where land tenure is insecure. This highlights the need for an appropriate institutional and regulatory environment to support implementation of restoration activities, to maximize local benefits.

Box 11.4 Implications of REDD+ for forest landscape restoration

Lera Miles

At the 11th Conference of Parties (COP) of the UN Framework Convention on Climate Change (UNFCCC) in 2005, an agenda item on 'Reducing emissions from deforestation in developing countries and approaches to stimulate action' was introduced by Papua New Guinea and Costa Rica with the support of eight other developing countries. Since 2005, the scope expanded twice, first to 'Reducing Emissions from Deforestation and forest Degradation' (REDD), and later to also cover emissions and sequestration resulting from other forest activities. At COP 13 in 2007, Parties agreed to strengthen efforts on REDD (UNFCCC, 2007). Around this time, several international and bilateral funds were established to support developing countries in preparing for REDD.

At UNFCCC COP15 in 2009, the Copenhagen Accord, agreed by a subset of prominent countries (UNFCCC, 2009a) mentioned REDD. There was also a Decision on methodological guidance for REDD, which formally broadened the scope to include other forest activities, so that Parties were now discussing REDD+ (UNFCCC, 2009b). COP16 in 2010 made further progress, requesting that developing countries interested in REDD+ work on strategies, reference emissions levels, forest monitoring, and on specific listed social and environmental safeguards. It also adopted the Copenhagen Accord's vision of a global mean temperature rise limited to less

Box 11.4 (cont.)

than 2°C. At the time of publication, Parties are still discussing the post-2012 emissions targets required to realise this vision.

REDD+ encompasses the following list of activities (UNFCCC 2010):

a. Reducing emissions from deforestation;

b. Reducing emissions from forest degradation;

c. Conservation of forest carbon stocks;

d. Sustainable management of forest;

e. Enhancement of forest carbon stocks.

Hence, forest (landscape) restoration, afforestation and reforestation activities could be eligible under the category 'enhancement of forest carbon stocks'.

There is common agreement that a developing country will be recompensed for REDD+ activities based on changes to its forest carbon balance, but no decision on whether the financial mechanism will be market- or fund-based. It is still to be negotiated whether the different activities (a) to (e) will be reported on and compensated in the same manner as one another. Some Parties wish to retain a distinction between activities that reduce emissions, and those such as forest restoration that sequester carbon; perhaps with REDD activities entering a carbon market, but conservation, sustainable management and enhancement relying on a fund. Others would prefer to see a full forest carbon accounting approach on a national level, with a single method and source of finance.

As the negotiations proceed, the Convention has requested countries to undertake preparatory and pilot work towards REDD+, on a voluntary basis. Finance for these activities has been made available to selected countries through the Forest Carbon Partnership Facility housed at the World Bank (FCPF), the UN-REDD Programme, and various bilateral agreements. Site and subnational pilot projects are also being funded by NGOs and the private sector.

Most of the REDD+ funds that are currently being made available through the FCPF, UN-REDD Programme and other sources are intended to foster 'REDD-readiness': that is, to support the development of national strategies for REDD+ implementation, pilot ("demonstration") projects and carbon monitoring systems.

The readiness strategies cover the appropriate use of carbon finance to achieve emissions reductions, and sometimes the attainment of additional benefits such as biodiversity.

Table 1 National participation of ReForLan countries in REDD readiness mechanisms (UN-REDD Programme, 2009; World Bank, 2009).

Country	Forest Carbon Partnership Facility (FCPF)	UN-REDD Programme
Argentina	Yes – accepted Oct 2008	Joined October 2009
Chile	Yes – accepted March 2009	No formal engagement
Mexico	Yes –accepted July 2008	Joined February 2010

All of the ReForLan countries (Argentina, Chile and Mexico) are among the 37 developing countries accepted for support by the FCPF (Table 1). Mexico was the first to join, and the most advanced in its progress through the World Bank process. FCPF has a two-phased approach, with the current Readiness Mechanism phase to be followed by the setup of a Carbon Finance Mechanism. Each country has submitted a Readiness Plan Idea Note (R-PIN) that sketches out its

Box 11.4 (cont.)

proposed activities under the first phase. Argentina and Mexico have also submitted a Readiness Preparation Proposal (R-PP), for fuller funding. FCPF does envisage funding reforestation and restoration activities, and its goals include testing ways to sustain or enhance the livelihoods of local communities and to conserve biodiversity within the approach to REDD+.

Argentina and Mexico have also joined the UN-REDD Programme. Funding has not yet been allocated to enable full participation of countries like these that joined after the first UN-REDD Programme pilot phase, but several additional sources of funds for the Programme have recently been announced. The UN-REDD Programme aims to support the full range of REDD+ activities, and has a strong emphasis on the ”multiple benefits” of REDD such as biodiversity. Forest restoration activities should be viable within the Programme provided that they form part of the country's National Programme (NP). Of the three countries, Mexico seems most likely to be willing to envisage forest restoration efforts as part of its REDD+ programme. In its R-PIN, Mexico describes its Strategic Forestry Programme for 2000–2025, and a number of initiatives that already promote reforestation through commercial plantations and forest restoration. It states that 'reforestation and commercial plantations are expected to have an effect in the medium to long term', by providing an alternative source of forest products, and suggests 'mainstreaming ecosystem conservation and restoration in sectoral policies outside the forest sector, such as for agricultural or transportation'. Argentina's R-PIN for FCPF mentions its National Programme for Native Forests Protection, whose objectives include 'reforestation and restoration plans for degraded native forests', but does not explicitly include these measures as part of its proposed implementation of REDD+. Chile's R-PIN does not mention the issue.

Overall, the REDD+ landscape can be confusing, with emerging models ranging from a top-down approach, concentrating on changes to national forest policy and institutions, and a project-based approach, aiming to trial REDD+ measures before scaling up to a national basis. Forest restoration could form part of either approach, but is only likely to be a common REDD+ measure in regions or countries where the carbon benefits are likely to be more cost-effective than for equivalent reductions in deforestation. Countries that are still rapidly deforesting are unlikely to see forest restoration as an urgent objective, unless the deforestation is yielding very high value products, and so would be particularly costly to limit. The challenges for forest restoration practitioners as national REDD+ policy develops are likely to include:

- ensuring that the afforestation and reforestation efforts prioritize forest restoration over development of commercial plantations
- ensuring that the selection of new forest areas meets conservation as well as carbon objectives
- ensuring that reforestation methodologies, including species selection and ongoing site management, meet (or do not harm) local conservation objectives
- identifying opportunities to propose specific restoration projects with dual carbon and conservation objectives under the REDD+ framework
- demonstrating the value of the multiple benefits arising from ecological restoration to assist in each of the above.

A specific risk for dry forest ecosystems is that there is a perception that compared to tropical moist forests, they are not especially valuable for carbon storage, and thus not a target for REDD+ efforts. In the short-term, it would be valuable to identify, review and highlight existing estimates of carbon stocks in dry forest ecosystems in comparison to alternative land uses in these areas. In the longer term, more data on this topic is clearly needed, including carbon in both above and below-ground biomass, and in soil carbon to the extent that it is vulnerable to land-use change.

Box 11.4 (cont.)

There is also a knowledge gap on the likely response of carbon stocks in forests and alternative land uses to a warming climate. Site managers may have a better idea of the likely local responses to a certain degree of warming, for example through hydrological constraints to carbon sequestration. The related risks and opportunities are clearly site and climate dependent – in some sites, it may be possible to argue that forest restoration will further reduce emissions under a warming scenario in comparison with the performance of existing land-cover.

In conclusion, it is critical for conservation professionals to pay attention to the development of national negotiating positions and REDD+ policy frameworks. Unintentional consequences can easily arise from negotiators' focus on and understanding of carbon rather than on the full suite of ecosystem services.

(Adapted from Miles, 2011).

Some tentative recommendations for policy and practice

Previous chapters have presented a number of recommendations to support the practical implementation of FLR (Chapters 2–9), including the creation of an enabling policy environment (Chapter 10), based on the research results obtained. To close this chapter, some additional recommendations are provided below. These are not always supported by a robust evidence base, and are therefore characterized by a high degree of uncertainty. Rather, they reflect the concerns of the research partners involved in the ReForLan project, having emerged through discussion at one of the project workshops. They should therefore be viewed as tentative, but nevertheless are informed by the practical experiences gained during implementation of this research project.

- *Recognize the value of degraded dryland forest.* Even degraded forest provides valuable ecosystem services, and is of value to biodiversity. Degraded forests should be protected and restored. Once completely deforested, sites will be much more difficult and expensive to restore. Stakeholders should be encouraged to engage with such protection.

- *Increase connectivity of dryland forest*. Dry forests have been highly fragmented by human activities, and this is likely to impair their ecological functioning and resilience. Future restoration efforts should seek to maximize forest connectivity within a landscape. Prioritization of areas for restoration should consider this as the most important criterion, from an ecological standpoint.

- *Biodiversity is more than species richness*. Some forest areas have low species richness, but are still of high value from a biodiversity conservation perspective. The presence of threatened, valuable or endemic species should be used to prioritize forests for restoration efforts, even if overall species richness is low. Research results indicated that the value of ecosystem services provided by species-poor forests can be as high, or even higher, than those that are more species rich.

- *Reduce human disturbance in dryland forest areas*. Restoration is less likely to succeed in disturbed sites. To increase the success of restoration efforts, there is a need to reduce human disturbance. On the other hand, it is recognized that disturbance is beneficial for some species. So disturbance needs to be carefully managed, not completely prevented.

- *Restore and maintain heterogeneity*. If ecosystem diversity within a landscape is high, it is more likely that provision of ecosystem services will be higher, and therefore the benefits to people will be higher. There is a risk that restoration actions could actually reduce ecosystem heterogeneity at the landscape scale; this should be avoided.
- *Dry forests are particularly important for provision of ecosystem services*. Preliminary evidence has highlighted the value of dryland forests for services such as production of fuelwood, charcoal, forage for cattle, and production of non-timber forest products. The perspectives of local communities are an essential source of information regarding the value of such services. Further evidence is required to document these values in greater detail. Human livelihoods in dryland areas may be particularly vulnerable and marginal, and particularly dependent on services provided by native forests.
- *Topographic heterogeneity may confer resilience to climate change*. Drylands situated close to areas where there is topographic heterogeneity may have greater resilience to climate change, because of the greater potential for species to respond dynamically to changes in climatic conditions. Connectivity is also key in this context, in terms of providing migration pathways. FLR should therefore increase resilience to climate change.
- *Involve local communities.* The success of restoration practices depends on the engagement of local communities. How can this best be achieved? Financial incentives do not necessarily work. Rather, there is a need to know the communities and their needs. Their involvement as stakeholders in the planning process is therefore key. There needs to be recognition of the efforts already being made by some landowners to restore forests (e.g. through reduced taxes). Provision of credit to communities could be of value. It is important that any incentives or rewards are correctly targeted to ensure that they do not act as a perverse incentive (e.g. for competing land uses). If incentives are provided, they need to be targeted appropriately, and their effects should be monitored. The scientific community should be involved in monitoring restoration and auditing it.
- *Incorporate FLR in land-use planning*. FLR should be incorporated as one element of an integrated approach to land-use planning. The scientific community should provide guidance to such planning efforts. Forest restoration can be integrated with other land uses.
- *Monitor FLR initiatives*. In order to assess whether FLR is cost effective, there is a need to closely monitor the implementation of restoration initiatives. This requires the use of appropriate indicators. While some indicators have been proposed (**Box 11.5**), their further testing and validation is required.

Box 11.5 Indicators for monitoring the implementation of Forest Landscape Restoration initiatives

These indicators were identified during a project workshop, in the context of developing tools for monitoring the effectiveness of restoration actions. This list is tentative, and does not represent a consensus among all workshop participants. Further research is required to refine, test and validate the indicators presented here. It is noted that some of these indicators would be difficult to measure in practice, and therefore measurable proxies may need to be used as an alternative.

Forest structure

- Species richness and composition
- Spatial configuration and amount of forest cover
- Presence of threatened species
- Presence of keystone species
- Presence of umbrella species
- Presence of endemic species
- Demographic and genetic structure
- Genetic diversity

Forest function

- Water infiltration and cycling
- Soil quality
- Nutrient cycling
- Viable populations of indicator species
- Natural regeneration
- Disturbance regime
- Microclimate
- Habitat condition
- Plant-animal interactions
- Presence of pollinators
- Carbon storage
- Gene flow
- Inbreeding

Benefits of restoration

- State of people's livelihoods – income, health, education
- Water supply
- Flood protection
- Soil protection and fertility
- Provisioning services – timber, fuel, food, forage, fibres, medicines
- Air quality
- Value for ecotourism, recreation
- Cultural and spiritual values
- Environmental awareness

Conclusions

One of the objectives of scientific research is to identify generalizations. This is particularly challenging for dryland forests, as in each respect – disturbance regime, diversity patterns, structure and composition, socioeconomic and cultural context – our research has demonstrated the overriding importance of local context. Given this, approaches to forest landscape restoration need to be flexible, and should be adapted to the particular conditions and characteristics of each location where it is implemented.

While our research has documented high rates of loss and degradation of dry forest, and intense human disturbance in many remaining forest stands, we have also demonstrated their potential for recovery. Forest landscape restoration approaches can therefore play a positive role in the conservation and sustainable management of this globally important forest type. In addition, our research suggests that forest restoration can be cost-effective, at least in some situations, in that the increase in benefits provided to people can outweigh the costs incurred. This is consistent with recent reviews highlighting the ecological feasibility restoration (Jones and Schmitz, 2009) and its potential benefits for both biodiversity and provision of ecosystem services (Rey Benayas *et al.,* 2009).

Any restoration action will incur costs. These can be significant, but can potentially be addressed using novel funding mechanisms, such as schemes supporting payment for ecosystem services (PES). Future restoration efforts need to ensure that both costs and benefits are equitably distributed.

The dryland forests of Latin America are of global conservation importance, but are also of high value to local communities and other stakeholders. While restoration approaches can potentially make a positive contribution to sustainable development of dryland areas, greater recognition of the high value of remaining dry forest is required. Protection of remaining dry forest areas, even those that are degraded, is an urgent priority.

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