# 15 Restoration of Forest Ecosystems in Fragmented Landscapes of Temperate and Montane Tropical Latin America

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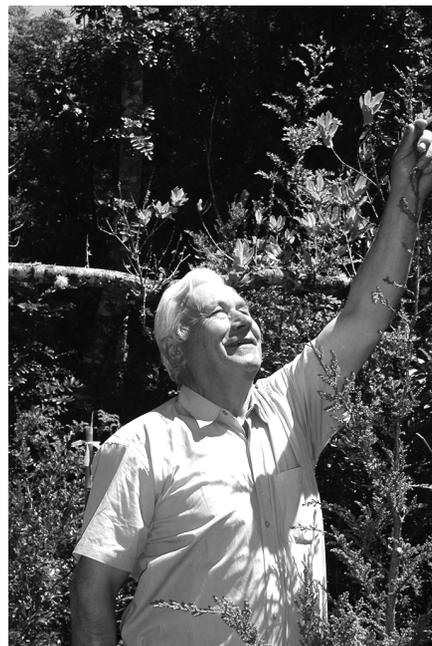
Photographs of Mr Alfredo Núñez illustrating the vegetation recovery and growth of *Fitzroya cupressoides* seedlings between 1998 and 2004 as part of an ecological restoration programme conducted by UACH researchers in Nuñez's property. Photos: Cristian Echeverría

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Summary

Temperate and tropical montane forests in Latin America represent a major natural resource at both regional and national levels for a number of reasons – biological, climatic, economic, cultural. Native tree species in these forests share conservation problems because of deforestation, habitat degradation, overall biodiversity loss and integrity of landscape structure. However, literature on forest restoration research and practices in these ecosystems is scanty and dispersed. We integrate forest restoration experiences aimed at a variety of purposes that allow us to gain insight over several years under contrasting ecological, social and economic conditions in six study regions: the Argentinian Andes, the IX and X Regions in Chile (including northern Chiloé Island), and central Veracruz and the central and northern Highlands of Chiapas (Mexico). By comparing analogous conditions and highlighting differences among the study sites, current pitfalls can be identified and used to define a minimum set of elements to be considered in a protocol for restoration practices. The restoration studies reviewed here include a wide variety of ecological and socio-economic circumstances that allow the identification of broad guidelines, criteria and indicators for planning, implementing and monitoring ecological restoration programmes. We conclude with statements that suggest approaches, strategies and concrete actions that might be considered as lessons learned and inputs for best practice in forest restoration research and programmes conducted in other developing regions.

Introduction

Temperate or tropical montane habitats occur in densely populated areas of most Latin American and Caribbean countries. These forests are not the most extensive types of forest ecosystems in Latin America, but their biodiversity, endemism and conservation threats are unusually high (Rzedowski, 1978, 1993; Donoso-Zegers, 1993; Hamilton *et al.*, 1995; Webster, 1995; Brown and Kappelle, 2001; Kappelle, 2004, 2006). The temperate and mountain forestlands represent a major natural resource at both regional and national levels for a number of reasons (biological, climatic, economic and cultural). In addition to their remarkable biological diversity, these forest communities are embedded within very different development contexts that must be considered in restoration programmes aimed at their sustainable use.

The temperate Andean forests of Chile and Argentina constitute a biogeographically isolated biome along both slopes of the Andes Cordillera, surrounded by the Pacific Ocean, the central Chilean Mediterranean scrub and the Atacama Desert farther north, the vast treeless semidesert and humid steppes and *pampas* east of the Cordillera, and subantarctic habitats in the southernmost lowlands of the continent (Cabrera and Willink, 1973; Armesto *et al.*, 1997). As observed in other temperate forest ecosystems of the world (broadly defined as those located at latitudes >30° either N or S of the equator), these forests have a relatively high productivity and show high regeneration dynamics (Donoso-Zegers, 1993; Donoso and Lara, 1998). However, these southern forests harbour more plant forms than their northern hemisphere counterparts, and a high level of endemism of vascular plants is one of their most striking attributes (e.g. 34% of the angiosperm genera; Armesto *et al.*, 1997).

The mountains of Veracruz and Chiapas (eastern and southern Mexico, respectively) include a number of highly diverse forest formations (Gómez-Pompa, 1973; Rzedowski, 1978; Breedlove, 1981; González-Espinosa *et al.*, 2004, 2005), from highly seasonal pine forest and pine–oak forest to formations such as montane rainforest (800–2500 m elevation) and evergreen cloud forest (>2500 m; Breedlove, 1981; Ramírez-Marcial, 2001; González-Espinosa *et al.*, 2006). The seasonal formations of Chiapas extend over a rather continuous distribution in the mountain systems of Guatemala, El Salvador, Honduras and northern Nicaragua (Kappelle, 2006). The optimal formations have a highly patchy distribution from subtropical areas in southern Tamaulipas through the Central American mountain ranges and the northern Andes, and are related in the south to the subtropical *Yungas* forests of southern Bolivia and north-east Argentina (Puig and Bracho, 1987; Brown and Grau, 1995; Hamilton *et al.*, 1995; Brown and Kappelle, 2001; Luna *et al.*, 2001). These forests harbour an outstandingly high biodiversity and contribute significant local inputs of water through fog condensation. Although it is recognized that they have a relatively poor primary productivity (Silver *et al.*, 2001), a considerable number of timber and non-timber products are obtained by local people, notably fuelwood (Brown and Kappelle, 2001).

Forest ecosystems represent a most valuable resource for people inhabiting the above-mentioned regions. Yet different social and economic contexts define distinct problems for conservation, sustainable use and restoration of their forest ecosystems. Rural communities in the mountains of Chiapas have some of the lowest well-being indices within Mexico, and their forest resources are currently used by a large part of the local population to provide them with non-commercial timber and firewood (Montoya-Gómez, 1998; Montoya-Gómez *et al.*, 2003). In contrast, in central Veracruz a mid-class group of landholders has become increasingly aware about the long-term benefits of conserving isolated remnant forest fragments for the provision of ecosystem services (Manson, 2004). In Chile forestlands are subjected to intensive management and provide forest products for global markets (Lara, 2004). Yet only 10% of the total rural communities in the country participate in this forestry industry, primarily involving those living where industrial plantations of exotic species have been promoted. Furthermore, many of these communities are among the most marginalized in Chile and have poverty indicators that have more than tripled in comparison to people living in urban regions (Sánchez *et al.*, 2002). In all countries here considered an overall legal framework is available to ensure the conservation and sustainable use of forests; yet they display considerable differences: law enforcement is still badly needed in southern Mexico, while in Chile a second-generation legislation process is currently under way in the Congress to protect native forests in particular (Lara, 2004).

Forests in these regions share a number of threats for the conservation of viable populations of native tree species and their sustainable use, including deforestation, habitat degradation, overall biodiversity loss and integrity of the landscape structure (Aldrich *et al.*, 1997; Ramírez-Marcial *et al.*, 2001, 2005; Galindo-Jaimes *et al.*, 2002; Williams-Linera, 2002; Newton *et al.*, 2004; Cayuela *et al.*, 2005, 2006a, b; and others in this volume). Native forest cover in the VII Region of central Chile has been reduced by 67% between 1975 and

2000, at an annual forest loss rate of 4.5%; corresponding figures for the more southern X Region during the same period are 24% of forest cover at an annual rate of 1.2% (see Chapter 2; Echeverría, 2005). In the VII Region, during the last three decades the native forest area has been mostly converted into forest plantations of exotic species, such as pines and eucalypts. In the X Region, loss of native forest has been associated with an expansion of agricultural land and forest logging for firewood and woodchips (Echeverría, 2005). In the central highlands of Chiapas deforestation has also been intense, but highly variable during the last three decades (de Jong *et al*., 1999; Ochoa-Gaona and González-Espinosa, 2000); annual deforestation rates ranged from 0.46% up to 3.42%. However, estimates for the last decade, which includes the start of the Zapatista revolt in 1994, indicate considerably higher rates: up to 4.98% (Ochoa-Gaona and González-Espinosa, 2000), and even higher than 6% (Cayuela *et al*., 2005). Nevertheless, loss of forest cover does not account for structural and floristic impoverishment in the remaining forest patches, which has also been considerable (González-Espinosa *et al*., 1995, 2006; Ramírez-Marcial *et al*., 2001; Galindo-Jaimes *et al*., 2002; Williams-Linera, 2002; Ochoa-Gaona *et al*., 2004; Chapter 3).

These considerations led us to conclude that forest restoration projects are badly needed in Latin America. Yet it should be recognized that a number of forest restoration initiatives have been undertaken. Furthermore, forest restoration projects in the region may represent some of the oldest (e.g. Janzen, 1987, 2002) or most ambitious in extent worldwide (e.g. Kageyama and Gandara, 2000; Wuethrich, 2007). Yet tropical lowland forests, mainly rainforests, have received most of the attention with respect to restoration projects in the region (Guariguata *et al*., 1995; Kageyama and Gandara, 2000; Janzen, 2002; Meli, 2003). In most cases the focus has been on the recovery of degraded rainforest stands; in other cases the establishment of selected tree populations, or forest cover, in old pasture or agricultural lands (Guevara *et al*., 1986, 1992; Aide *et al*., 2000; Janzen, 2002; Florentine and Westbrooke, 2004). Much emphasis has been placed on the role of vertebrates (including domesticated animals; Posada *et al*., 2000) in seed dispersal from naturally established standing remnant trees (e.g. Otero-Arnáiz *et al*., 1999; Toh *et al*., 1999; Cubiña and Aide, 2001). Less common have been efforts involving enrichment planting in stands with degraded floristic, structural and functional attributes (e.g. Ramos and del Amo, 1992; Montagnini *et al*., 1995).

In this chapter, we draw upon forest restoration experiences aimed at a variety of purposes pursued for several years under contrasting ecological, social and economic conditions in six temperate or tropical mountain study regions located in Argentina, Chile and Mexico. By comparing analogous conditions or stressing differences among the study sites we suggest approaches, strategies and concrete actions that might be considered as lessons learned and best practice in forest restoration. Starting from the discussion of results obtained, we aim to identify general issues that might offer insights for planning, implementing and monitoring restoration programmes in other developing regions that share socio-economic and natural attributes with our study sites.

Definition and Description of Forest Restoration

Forest restoration in our study sites may potentially include a variety of practices and purposes, but two have been more frequently defined as a goal: (i) establishment of native tree species in open areas, frequently after agricultural use; and (ii) floristic enrichment of impoverished secondary stands, frequently after selective logging of timber trees and saplings for firewood. We adopt the concept that forest restoration should be defined broadly, with an aim towards the eventual attainment of environmental health as indicated by forest structure, floristic composition and ecosystem functioning, along with social and financial viability of forest utilization. In the long term, we propose for our study regions that restoration of forest habitats should aim to support the enhancement of a respectful attitude towards nature and culture, social welfare, political coexistence and tolerance, and aesthetic and historical values, among others (Higgs, 1997; Cairns, 2002; González-Espinosa *et al.*, 2007).

Forest restoration practices attempt to simulate ecological processes influential during secondary succession (Bradshaw, 1987, 2002). In each particular case study, the practices we have used follow different approaches to simulate mechanisms of succession. In the central Highlands of Chiapas and central Veracruz, a major concern has been the utilization of a large number of native tree species in order to restore the high local diversity. This approach has required the experimental study of germination requirements and response of seedlings and juveniles of key species to gradients of shade and temperature (Alvarez-Aquino *et al.*, 2004; Ramírez-Marcial *et al.*, 2005). Although restrictions on genetic variation imposed by secondary forest regeneration in highly diverse forests are recognized (Sezen *et al.*, 2005), this issue has not yet been a major concern in our Mexican studies (but see Rowden *et al.*, 2004). On the other hand, in the Chilean and Argentinian sites, interest has concentrated on threatened endemic conifer species. Species have been investigated singly, and emphasis has been given to conserving the genetic variation of highly threatened populations (Premoli *et al.*, 2001, 2003; Bekessy *et al.*, 2002; Allnutt *et al.*, 2003) and to identify particular environmental factors limiting recruitment and establishment (e.g. seed dispersal, water-table fluctuations).

Study Regions

The 33 sites within the six study regions encompass a considerable range of ecological conditions in areas close to the northern limits of the mountain cloud forests (Williams-Linera, 2002) down to the central distribution of the South American temperate forests in northern Chiloé Island (Armesto *et al.*, 1998). An envirogram plotting the values of mean annual rainfall and mean annual temperature for the 33 study sites in all regions indicates that – with the possible exception of very dry and cold sites – most of the combinations between 1000 and 2200 mm of annual rainfall and 8°C and 22°C of mean annual temperature have been included in our restoration essays (Fig. 15.1). The South American sites are
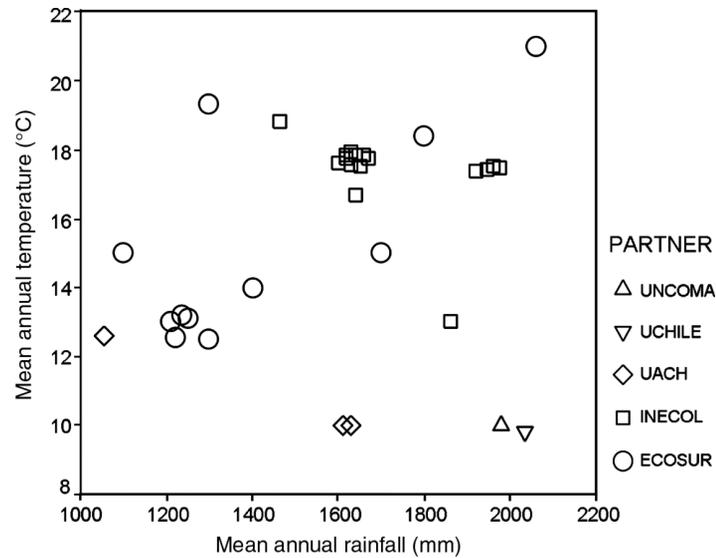
Fig. 15.1. Scatter plot of mean annual rainfall (MAR, mm year^{-1}) and mean annual temperature (MAT, °C) for the 32 field sites where the BIOCORES project partners have established forest restoration essays. UNCOMA, Universidad Nacional del Comahue (Bariloche, Argentina); UCHILE, Universidad de Chile (Chiloé Island, Chile); UACH, Universidad Austral de Chile (IX and X Regions, Chile); INECOL, Instituto de Ecología (central Veracruz, Mexico); ECOSUR, El Colegio de la Frontera Sur (central and northern Highlands of Chiapas, Mexico). See Tables 15.1 and 15.2 for additional details.

within a belt of cold temperatures (10–13°C) at relatively low elevations, and represent a set of low-energy sites (annual actual evapotranspiration, *AAET*, mostly lower than 600 mm year^{-1}; Table 15.1). Most of the sites in central Veracruz are located in habitats within a very narrow belt of mean annual temperatures and an annual rainfall range of *c.*1500 mm. Finally, the Chiapas sites include the widest range of probed environmental conditions, including the warmest and wettest sites among the whole set. Moreover, estimates of biologically useful energy (Rosenzweig, 1968) in Chiapas sites range from *c.*1000 up to >2100 mm year^{-1} and facilitate comparisons among all the study sites (Table 15.1).

Case Studies

Argentina (Site 1)

Restoration trials with Nothofagus pumilio *(Lenga)*
We established a long-term reciprocal transplant experiment to compare seedling growth (height, basal diameter and architecture) in 220 plants from two elevations: 1500 and 1000 m (Table 15.2). Seedlings were planted at both

Table 15.1. Location name, geographical coordinates, mean elevation (m), mean annual temperature (*MAT*, °C), mean annual rainfall (*MAR*, mm year^{-1}), annual actual evapotranspiration (*AAET*, mm year^{-1}; estimated with model by Turc (1954)), predominant landform and soil types/attributes of the study sites.

Partner	Site no.	Name of site	Latitude	Longitude	Elevation	MAT	MAR	AAET	Landform	Soil type/attributes
UNCOMA	1	Nahuel Huapi National Park	41° 08' S	71° 19' W	1000–1500	10.0	1980	577	Hillside	Andosols
UCHILE	2	Senda Darwin	45° 53' S	73° 40' W	40	9.8	2035	571	Flat area	Ñadi type (shallow flooded peat bog) on Fe/Si duripans
UACH	3	Villa Las Araucarias	38° 30' S	73° 16' W	630	12.6	1055	601	Flat area, hillside, steep slope	Shallow fertile derived from granite
UACH	4	Fundo Núñez	41° 26' S	73° 07' W	65	10.0	1610	566	Flat area	Ñadi type (shallow flooded peat bog) on Fe/Si duripans
UACH	5	Lahuen Ñadi	41° 26' S	73° 07' W	62	10.0	1630	566	Flat area	Same as in Site 4
INECOL	6	Rancho Viejo	19° 30' N	97° 00' W	1500	17.8	1650	884	Gentle and steep slopes	Acrisols, with high or very high organic matter content
INECOL	7	Xolostla 1	19° 32' N	96° 58' W	1450	17.8	1650	884	Same as above	Same as above
INECOL	8	Xolostla 2	19° 32' N	96° 58' W	1450	17.8	1650	884	Same as above	Same as above
INECOL	9	Xolostla 3	19° 32' N	96° 58' W	1450	17.8	1650	884	Same as above	Same as above
INECOL	10	Rancho Raúl	19° 31' N	96° 58' W		17.8	1650	884	Gentle slopes	Same as above
INECOL	11	Nicoletta	19° 32' N	96° 58' W		17.8	1650	884	Gentle slopes	Same as above
INECOL	12	Casazzas	19° 32' N	96° 58' W		17.8	1650	884	Gentle slopes	Same as above

Table 15.1. *Continued*

Partner	Site no.	Name of site	Latitude	Longitude	Elevation	MAT	MAR	AAET	Landform	Soil type/attributes
INECOL	13	El Cedro	19° 32' N	96° 58' W		17.8	1650	884	Gentle slopes	Same as above
INECOL	14	Capulines	19° 31' N	96° 59' W		17.8	1650	884	Gentle slopes	Same as above
INECOL	15	Rancho Olinca	19° 32' N	96° 59' W		17.8	1650	884	Gentle slopes	Same as above
INECOL	17	Las Cañadas 2	19° 11' N	96° 59' W	1340	17.4	1960	899	Same as above	Same as above
INECOL	18	Las Cañadas 3	19° 11' N	96° 59' W	1340	17.4	1960	899	Same as above	Same as above
INECOL	19	Las Cañadas 4	19° 11' N	96° 59' W	1340	17.4	1960	899	Same as above	Same as above
INECOL	20	La Martinica	19° 35' N	96° 57' W	1470	18.8	1460	896	Gentle slopes	Same as above
INECOL	21	Mesa de La Yerba	19° 33' N	97° 01' W	1875	13.0	1860	688	Gentle slopes	Same as above
ECOSUR	22	Rancho Merced-Bazom 1	16° 44' N	92° 29' W	2350	13.0	1250	642	Flat area, gentle slopes	Luvisol, rendzina
ECOSUR	23	Rancho Merced-Bazom 2	16° 44' N	92° 29' W	2300	13.0	1250	642	Flat area, steep slopes	Luvisol, rendzina
ECOSUR	24	Rancho Merced-Bazom 3	16° 44' N	92° 29' W	2350	13.0	1250	642	Flat areas	Luvisol, rendzina

ECOSUR	25	Rancho Merced-Bazom 4	16° 44' N	92° 29' W	2400	13.0	1280	642	Flat areas	Luvisol, rendzina
ECOSUR	26	Corazón de María	16° 41' N	92° 32' W	2380	15.0	1100	682	Flat areas	Vertic cambisol, gleysol
ECOSUR	27	Universidad Lindavista	17° 10' N	92° 54' W	1720	15.0	1700	763	Steep slope	Luvisol, lithosol, rendzina
ECOSUR	28	Estación Biológica Huitepec	16° 44' N	92° 41' W	2500	12.5	1300	631	Gentle and steep slopes	Cambisol, acrisol
ECOSUR	29	Moxviquil	16° 45' N	92° 38' W	2130	13.0	1200	635	Steep slopes	Lithosol
ECOSUR	30	Mitzitón	16° 40' N	92° 33' W	2400	14.0	1400	695	Flat area, gentle slopes	Luvisol, rendzina
ECOSUR	31	San Cayetano	16° 57' N	92° 46' W	1620	18.4	1800	933	Steep slopes	Phaeozem
ECOSUR	32	La Trinitaria	16° 08' N	92° 04' W	1590	19.3	1300	877	Flat	Vertisol
ECOSUR	33	Montebello	16° 04' N	91° 37' W	1520	21.0	2060	1,108	Flat and gentle slopes	Luvisol, rendzina

UNCOMA, Universidad Nacional del Comahue (Bariloche, Argentina); UCHILE, Universidad de Chile (Chiloé Island, Chile); UACH, Universidad Austral de Chile (IX and X Regions, Chile); INECOL, Instituto de Ecología (Xalapa, Mexico); ECOSUR, El Colegio de la Frontera Sur (central and northern Highlands of Chiapas, Mexico).

Table 15.2. Forest field restoration experiments conducted in 33 study sites. Months of duration up to May 2005 in each of the study sites in all regions, plot sizes, number of species included, initial condition being restored (AP, abandoned pasture; BF, bog field; DF, degraded forest; ESF, early secondary forest; FE, forest edges; FF, fallow field; FI, recent fire; MSF, mid-successional forest; OA, open area; OF, old-growth forest; SH, shrubland), plant performance variables measured (G, % germination; R, natural recruitment; S, % survival; H, stem height; B, basal stem diameter), conclusions on possible mechanisms or ecological interactions supposed to be implied, and identity of studied species. Numbers in parentheses within the same site (see Table 15.1) refer to particular studies in the site as indicated in the text.

Site	Months	No. plots	Plot size (m²)	No. species	No. plants	Initial condition	Variables measured	Conclusions on mechanisms or interactions involved	Species included
1 (1)	12	1	120	1	220	DF	S, H, B, architecture	Adaptation to contrasting elevations	*Nothofagus pumilio*
1 (2)	12	1	50,000	1	3,000	DF, FI	R, S, H, B	Facilitation by shrubs and herbs after fire	*Austrocedrus chilensis*
2 (1)	33	4	< 2,500	1	392	BF, FI, OA, SH	S, H	Interference by *Sphagnum* sp. moss	*Pilgerodendron uviferum*
2 (2)	24	1	5,000	4	?	AP	R	Increased seed density by perching birds	*Amomyrtus luma, Berberis buxifolia, Berberis darwinii, Drimys winteri*
3 (1)	10	–	–	1	800 seeds	OA, SH	G	Seed stratification at 4°C	*Araucaria araucana*
3 (2)	24	2	10,000 & 5,000	1	200	OA, SH	R, S, H	Cyclical seed production	*Araucaria araucana*
3 (3)	8	4 (?)	?	1	?	BF, DF	S, H	Effects of root pruning and mycorrhization	*Araucaria araucana*
4 (1)	68	1	2,650	1	700	AP, BF	S, H, B	Negative effect of drainage on plant performance	*Fitzroya cupressoides*
5 (1)	32	1	?	1	1,076	AP, BF	S, H, B	Negative effect of drainage on plant performance	*Fitzroya cupressoides*

6–9, 16–19 (1)	70, 30	9	?	7	1,680	ESF, DF, FF	S, H, B	Identification of functional groups: light demanding, shade tolerant, intermediate	*Carpinus caroliniana, Fagus grandifolia var. mexicana, Juglans pyriformis, Liquidambar styraciflua, Podocarpus matudai, Quercus acutifolia, Symplocos coccinea*
10–15, 20, 21 (2)	18	6	?	6	?	AP, FE, FF	S, H, B	Competition with grasses, shading by established trees, underground herbivory by pocket gophers (*Thomomys* sp.)	*Fagus grandifolia var. mexicana, Quercus germana, Quercus xalapensis, Trema micrantha, Heliocarpus donnell-smithii, Rapanea myricoides*
22 (4)	54	6	100	9	486	DF, FE, FF	R, S, H, B	Facilitation by light-demanding species is not a requisite for enrichment of secondary stands	*Arbutus xalapensis, Clethra pachecoana, Cornus disciflora, Olmediella betschleriana, Prunus rhamnoides, Prunus serotina ssp. capuli, Quercus crassifolia*
23	60	2	2,500	11	?	Pine-dominated ESF	R, S, H, B	A pine-dominated canopy benefits broadleaved late successional species	*Acer negundo ssp. mexicana, Buddleja cordata, Liquidambar styraciflua, Magnolia sharpii, Photinia microcarpa, Prunus lundelliana, Quercus crispipilis, Quercus laurina, Quercus rugosa, Styrax magnus, Ternstroemia lineata ssp. chalicophyla*
24	48	6	2,100	5	1,470	DF	S, H, B	Different responses of oak species across the forest-edge–grassland gradient	*Quercus candicans, Quercus crassifolia, Quercus laurina, Quercus rugosa, Quercus segoviensis*

Continued

Table 15.2. *Continued*

Site	Months	No. plots	Plot size (m²)	No. species	No. plants	Initial condition	Variables measured	Conclusions on mechanisms or interactions involved	Species included
25	72	8	Variable, mostly c.400	5	596	MSF, OF	S, H, G	Plant performance depends on relative conditions of the light environment in addition to point-level values	*Alnus acuminata* ssp. *arguta, Cornus excelsa, Liquidambar styraciflua, Persea americana, Quercus laurina*
26	120	8	400	4	1,656	OA, SH	R, S, H, B	Dominant shrub *Baccharis vacci-nioides* functions as a nurse plant for trees	*Pinus ayacahuite, Pinus pseudostrobus* var. *apulcensis, Quercus crassifolia, Quercus rugosa*
27	48	4	1,800	16		OA	S, H, B	Facilitation is possible; a requisite for restoration of open areas	*Acer negundo* ssp. *mexicana, Clethra pachecoana, Cleyera theaeoides, Cornus disciflora, Liquidambar styraciflua, Magnolia sharpii, Oreopanax xalapensis, Quercus candicans, Pinus chiapensis, Podocarpus matudai, Prunus rhamnoides, Psychotria galeottiana, Zanthoxylum melanostictum, Styrax magnus, Symplocos limoncillo*
28	96(180)	6	1,000	7		AP/SH, MSF, OF	S, H, B	Conifers perform well in open areas; broad-leaved understorey tree species require facilitation by other species providing partial shade	*Abies guatemalensis, Oreopanax xalapensis, Pinus ayacahuite, Pinus pseudostrobus* var. *apulcensis, Rhamnus sharpii, Ternstroemia lineata* ssp. *chalicophyla*

29	60	4	1,500	25	OA	R, S, H, B	Facilitation is possible; required for restoration of open areas	Acer negundo ssp. mexicana, Alnus acuminata ssp. arguta, Arbutus xalapensis, Buddleja cordata, Chiranthodendron pentadactylon, Clethra pachecoana, Cleyera theaeoides, Cornus disciflora, Cornus excelsa, Ehretia thinifolia, Ilex vomitoria, Liquidambar styraciflua, Olmediella betschleriana, Persea americana, Pinus pseudostrobus var. apulcensis, Psychotria galeottiana, Prunus brachybotria, Prunus rhamnoides, Prunus serotina ssp. capuli, Quercus crispipilis, Quercus rugosa, Rapanea juergensenii, Rhamnus sharpii, Styrax magnus, Zanthoxylum melanostictum
30 (4)	54	4	100	9	FE, FF, DIF	R, S, H, B	Facilitation by light-demanding species is not a requisite for enrichment of secondary stands	Arbutus xalapensis, Clethra pachecoana, Cornus disciflora, Olmediella betschleriana, Prunus rhamnoides, Prunus serotina ssp. capuli, Quercus crassifolia, Quercus laurina, Quercus rugosa
30 (6)	22	21	100	10	AP, SH	S, H, B	Above- and below-ground competition with grasses; different between grasslands and shrublands. Facilitation of grass cover on seedlings observed in a few cases	Alnus acuminata ssp. arguta, Garrya laurifolia, Nyssa sylvatica, Pinus ayacahuite, Pinus pseudostrobus var. apulcensis, Pinus serotina ssp. capuli, Quercus crispipilis, Rapanea juergensenii, Styrax magnus, Ternstroemia lineata ssp. chalicophyla

Continued

Table 15.2. *Continued*

Site	Months	No. plots	Plot size (m²)	No. species	No. plants	Initial condition	Variables measured	Conclusions on mechanisms or interactions involved	Species included
31	12	1	500	16	200	OA	S, H, B	Facilitation is possible; a requisite for restoration of open areas	*Cornus excelsa, Fraxinus uhdei, Juniperus gamboana, Olmediella betschleriana, Pinus ayacahuite, Pinus montezumae, Prunus serotina ssp. capuli, Quercus crassifolia, Quercus crispipilis, Quercus rugosa, Quercus sapotifolia, Quercus segoviensis, Quercus sp., Randia aculeata, Turpinia tricornuta*
32	9	1	400	16	1,032	OA	S, H, B	Facilitation is possible and a requisite for restoration of open areas	Same as above
33	21	8	2,500	16	3,200	FI, ESF, OA, SH	R, S, H, B	Facilitation is possible and a requisite for restoration of open areas	*Ilex vomitoria, Myrica cerifera, Olmediella betscheriana, Oreopanax xalapensis, Prunus brachybotria, Prunus lundelliana, Psychotria galeottiana, Quercus sapotifolia, Quercus sp., Randia aculeata, Rapanea myricoides, Rhamnus capraeifolia, Styrax magnus, Synardisia venosa, Turpinia tricornuta*

elevations in Chall-Huaco Valley, Nahuel Huapi National Park in May 2005. Previous studies indicate that individuals from subalpine contrasting elevations may be genetically different due to reproductive barriers to gene flow exerted by phenological differences (Premoli, 2003). Furthermore, greenhouse experiments have shown heritable variation in ecophysiological traits along with morphological and phenological differences associated with elevation (Premoli, 2004). Results of the reciprocal transplant experiments will allow the testing of adaptive differences between plants from different provenances that will guide restoration trials.

Restoration trial with Austrocedrus chilensis *(Ciprés de la Cordillera)*
A restoration essay was established on *c*.5 ha of hillside originally covered by monospecific *Austrocedrus chilensis* forest near the Nahuel Huapi National Park. The entire area was burnt four years before the start of the study and then illegally logged. *Austrocedrus* is affected by fire and herbivore browsing, and early regeneration stages are highly dependent on facilitating shrubs (Kitzberger *et al.*, 2000; Rovere *et al.*, 2005). Various interest groups are participating in the study including: (i) the private sector, represented by a company that provides the study site; (ii) the Provincial Government, represented by Servicio Forestal de la Provincia de Río Negro (Río Negro Province Forest Service), which supplies plants and provides logistic support; and (iii) Universidad Nacional del Comahue, responsible for designing and monitoring the study, as well as for organizing activities aimed at environmental education in the local community. Vegetation and forest floor cover, and natural regeneration of *A. chilensis* were initially assessed. We planted 3000 trees during winter 2004 (Table 15.2). Preliminary results indicate that shrub cover after fire is high (54%). Natural regeneration of *A. chilensis* has been very low (less than one sapling per ha), but preliminary results indicate that survival and establishment are facilitated by shrubs and herbs.

Chile: Northern Chilóe Island (Site 2)

Long-term restoration of Pilgerodendron uviferum *(Ciprés de las Guaitecas)*
The experiment was established in August 2002, in an open area that was subjected to a fire and became wet shrubland afterwards (Table 15.2). Little regeneration and slow succession are currently observed. Seasonal flooding caused by logging and burning of the forest favours invasion by *Sphagnum*. The study assesses the effects of the substrate of *Sphagnum* moss on growth and survival of *Pilgerodendron uviferum* in areas disturbed by human impact. The experiment includes two sites with four plots in each within a multifactorial design; plants were spaced at 1 m distance ($N = 49$ in each plot). The plants were obtained from cuttings and grown for two years in the nursery at Senda Darwin Biological Station. Plants of different origins and known gender were randomly allocated among plots. The sites were with and without *Sphagnum*. Growth of *P. uviferum* was similar during the first years of the study, yet plant responses were significantly different
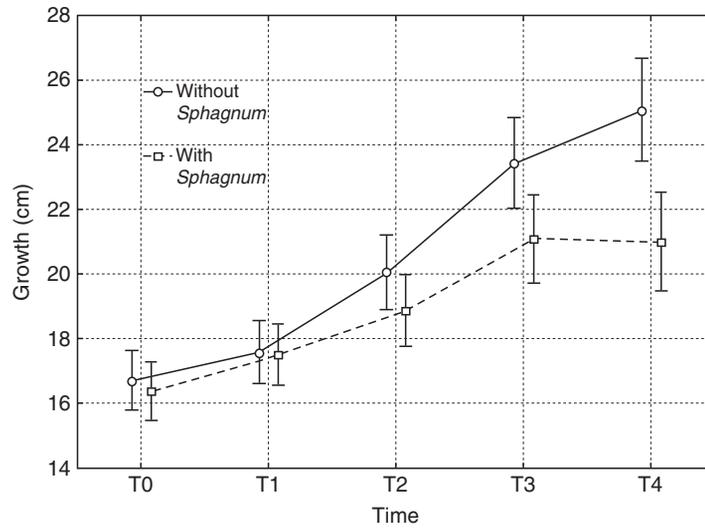
Fig. 15.2. Growth response of *Pilgerodendron uviferum* in plots with and without *Sphagnum* sp. moss at Senda Darwin Biological Station, northern Chiloé Island, Chile. T0 is August 2002; T4 is February 2005.

by early 2005: saplings in plots without *Sphagnum* grew more than those in plots with *Sphagnum* (Fig. 15.2). A repeated measures ANOVA on $\log_{10}$ growth showed significant interaction between substrate and time ($P < 0.001$). However, per cent survival was not significantly different in plots with *Sphagnum* treatments. The preliminary results suggest that *Sphagnum* cover seems to have a negative effect on growth of *P. uviferum*; so far survival seems to be unrelated to substrate.

Effects of coarse woody debris and bird perches on tree recruitment in artificial prairies

A number of studies in the temperate rainforest of Chiloé Island show that many trees, shrubs and vines display a bird-dispersal syndrome. It is also known that seed rain is much lower in shrublands and prairies than in forest fragments. This study aims to assess: (i) the effects of different substrates on the establishment of woody species in anthropogenic prairies; and (ii) the effect of artificial perches that could be used by birds in facilitating the establishment of bird-dispersed plants. Different substrates (logs, woody detritus of *Drimys winteri* (Canelo) and *Nothofagus dombeyi* (Coigüe común)) and prairie soil with or without perches were randomly distributed in artificial prairies at Senda Darwin Biological Station ($N = 180$). Seed deposition has only been observed on woody detritus and log substrates. To evaluate the function of perches, we sampled seed rain in traps with and without perches in the same artificial prairies. After four months, we found seeds in all traps with perches ($N = 15$) and only in eight traps without perches. The species found were *D. winteri*, *Amomyrtus luma* (Luma), *Berberis buxifolia* and *Berberis*

darwinii (all dispersed by birds; Table 15.2). Number of seeds per trap was different between perch and non-perch treatments ($P < 0.0001$). These data indicate that the presence of perches may increment the seed rain of bird-dispersed woody species in prairies of Chiloé Island.

Chile: Region IX (Site 3)

Activities have been conducted in two sites of the Cordillera de la Costa (Coastal Range): Villa Las Araucarias and Nahuelbuta National Park (Table 15.2). Tree cores ($N = 200$) and chunks ($N = 15$) collected at Villa Las Araucarias are being cross-dated to date the occurrence of fires and to generate a fire chronology of *Araucaria araucana*. However, cross-dating has been trouble-some because the trees are in flat areas and fires are highly frequent. Fire scars are produced on the trunk perimeter, and not in a particular area of the stem as in hilly areas. Only a few samples from Nahuelbuta National Park have been cross-dated due to the difficulty in differentiating the tree rings. Additional samples are currently being collected to obtain an improved fire chronology.

In March 2004 we collected seeds of *A. araucana* to produce plants for restoration and research activities. In October 2004 the seeds were sown using four different germination treatments (four replicates of 50 seeds each). High germination was observed in control seeds; the seeds were stored at 4°C from March through October, which could cause their stra-tification and therefore reduce the effect of the pre-germination treatments. Also, no control was implemented on treatment location inside the green-house; germination of the untreated seeds could be enhanced in the south side.

In 2003 two plantations of *A. araucana* from seeds collected at Villa Las Araucarias were established in two permanent plots of 1.0 and 0.5 ha (labelled as plots 1 and 2, $N = 100$ per plot). Survival and growth of seedlings and sap-lings were assessed in 30 and 20 subplots distributed within the two plots. Mortality of *A. araucana* plants in April 2005 was higher in plot 2 (25%) than in plot 1 (12%). These trends in mortality are similar to those recorded in March 2004 (17% and 20%, respectively). This variation in mortality rate between sites could be explained by differences in the site and canopy cover. Plot 1 is on a steep slope and has some canopy protection from remaining trees; plot 2 is a flat, open site. Scarce natural regeneration has been observed, most probably due to an extremely low production of seeds during the last two years in Nahuelbuta National Park and null in Villa Las Araucarias. Given the biannual seeding cycles of *A. araucana*, we anticipate higher seed production in 2006 and 2007.

In addition, in 2004 new plantations were established in sites with different levels of forest cover: Site A, a gap within a plantation of the exotic *Pinus radiata*; Site B, under the canopy of *P. radiata* trees; Site C, with side-protection by *N. dombeyi* and *A. araucana*; and Site D, a small depression covered by peat bog. The lowest and highest mortalities were obtained in sites D (4%) and A (8%).

These low mortality values are considered as quite favourable, given the extremely harsh climate of the study area. Currently, we are planning to improve the survival rates of *A. araucana* seedlings by applying cultural treatments such as root pruning, mycorrhization and *in situ* production of plants.

Chile: Region X (Sites 4 and 5)

Long-term restoration of Fitzroya cupressoides *(Alerce)*

Ecological restoration works have been conducted in two study areas in the X Region. The first plantation of *Fitzroya cupressoides* was at the property of Mr Alfredo Núñez (hereafter Fundo Núñez) in 1998 (Table 15.2). Plants were produced from seeds and cuttings collected in the local area. Monitoring activities such as assessments of survival and growth in height and diameter have been undertaken each year. In September 2002, another plantation was established at the Lahuen Ñadi Park with cuttings from a local population. Until April 2005, plant mortality at Fundo Núñez was 12%. Mean increase of stem height of *F. cupressoides* at Fundo Núñez has been 10.3 cm year^{-1} between 1999 and 2005. Yet, in well-drained areas within the plot, mean growth rate has been 31.8 cm year^{-1}. At Lahuen Ñadi, mean growth has been 4.4 cm year^{-1}. These marked differences may be explained by the drainage conditions where the plants are established, as most microsites at Lahuen Ñadi are poorly drained. A total of 160 seed traps were installed in June 2003 at Fundo Núñez to collect seeds of *F. cupressoides* as a function of wind direction. Seed production is highly variable among years: a total of 29,477 seeds were collected in 2003, but only 217 and 257 seeds in 2004 and 2005, most of them moved by winds with N–S or S–N orientation. To analyse the effect of water-table fluctuations on the establishment and growth of *F. cupressoides* plants, several piezometers have been installed (22.6 devices ha^{-1}) at Fundo Nuñez. Results have revealed that variations in plant growth have been associated with fluctuations in the water-table level.

Mexico: Central Veracruz (Xalapa; Sites 6–21)

Restoration of tropical montane cloud forests

A major goal of restoration activities in central Veracruz has been the maintenance of regional diversity. Since 1998 a number of tree restoration plots have been established and monitored to determine the potential of ecological restoration with selected native tree species and to define criteria for matching these species with particular microhabitat conditions. The native tree species used were *Carpinus caroliniana*, *Fagus grandifolia* var. *mexicana*, *Juglans pyriformis*, *Liquidambar styraciflua*, *Podocarpus matudai*, *Quercus acutifolia* and *Symplocos coccinea* (Table 15.2). The restoration experiments were conducted in three forest fragment interiors, three post-agriculture fallow fields adjacent to the forest fragments, and three early secondary forest stands (*acahuales*). Results were compared with on-farm plantations established by private landowners. Plant performance was evaluated as survival, and increment in stem height and basal

stem diameter. Other variables monitored were natural recruitment, soil pH, organic matter and compaction. Responses were integrated using functional groups (light-demanding, shade-tolerant and intermediate). Initial age and seedling height had a significant effect on survival, but not on height or diameter increment across all species and sites. Overall survival was highest in early secondary forests (70%), followed by forest interior (42%) and fallow field (36%). Maximum height was recorded outside the forest. Average stem height was greater in the adjacent agricultural fields (4.6 m) and in early secondary forests (3.6 m) than in the forest fragment interiors (0.62 m). Annual diameter increment rate was lower in forest interior (0.22 cm year^{-1} in 2000, and 0.04 cm year^{-1} in 2004) than in adjacent field (1.04 and 0.64 cm year^{-1}) and in old-field sites (0.66 and 0.50 cm year^{-1}). *Juglans*, *Podocarpus* and *Quercus* exhibited the greatest survival (62–80%), but intermediate relative growth rates in stem height (26–57 cm year^{-1}; Fig. 15.3); *Carpinus* and *Liquidambar* showed intermediate survival (50–54%), but high growth increments (45–96 cm year^{-1}); and *Fagus* and *Symplocos* displayed low survival (18–20%) and low height increments (13–29 cm year^{-1}). We conclude that performance of different tree species depends on specific level of disturbance exhibited at each site, suggesting the importance of accurate species–site matching to obtain optimum rates of survival and growth in particular scenarios. *Juglans* and *Quercus* have the potential to be used in the rehabilitation of degraded and disturbed areas, respectively; *Podocarpus* can be used in plantation enrichment; *Liquidambar* and *Carpinus* may be used to expand the extent of cloud forest; *Fagus* and *Symplocos* can survive and grow in forests other than those in which they are naturally present.

Forest restoration in abandoned pastures

Land clearing to establish pastures with non-native grasses and urban/suburban development has been a common practice in central Veracruz over the last 50 years. Yet opportunities to restore montane cloud forests from abandoned pastures exist as land use changes due to low productivity. We established six restoration plantations by planting seedlings of three primary
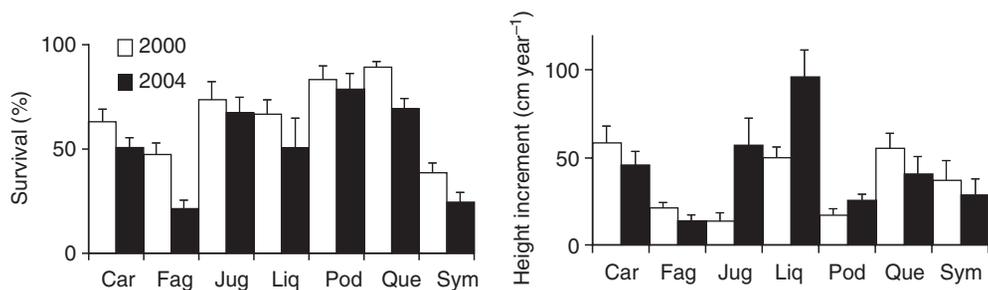

Fig. 15.3. Per cent survival between 2000 and 2004 and growth rates in stem height (cm year^{-1}) at 2000 and 2004 for native tree species used in restoration field experiments in central Veracruz, Mexico. Car, *Carpinus caroliniana*; Fag, *Fagus grandifolia* var. *mexicana*; Jug, *Juglans pyriformis*; Liq, *Liquidambar styraciflua*; Pod, *Podocarpus matudai*; Que, *Quercus acutifolia*; Sym, *Symplocos coccinea*.
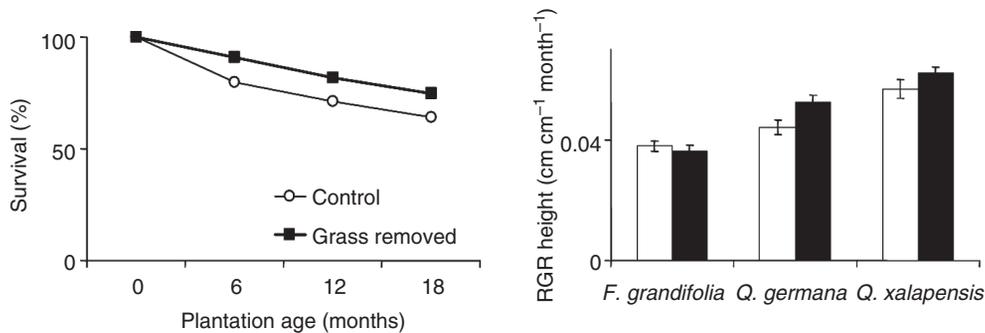
Fig. 15.4. Per cent survival of seedlings with and without surrounding grass cover and relative growth rates in stem height (cm cm^{-1} month^{-1}) of three primary tree species (*Fagus grandifolia* var. *mexicana*, *Quercus germana* and *Quercus xalapensis*) used in restoration experiments in abandoned pastures in central Veracruz, Mexico.

tree species (*Fagus grandifolia* var. *mexicana*, *Quercus germana* and *Q. xalapensis*) in three recently (<1 year) and three long-abandoned pastures (12–17 years); the seedlings were planted at 0, 10 and 40–50 m from the forest border. A treatment of removal of herbaceous vegetation was included. Sapling survival was higher when grasses (mostly the stoloniferous exotic *Pennisetum clandestinum*) were removed than in controls. All species attained larger diameter and height growth in plots with grass removed in comparison to controls (Fig. 15.4.). Survival of *F. grandifolia* and *Q. germana* was higher in older fields, while *Q. xalapensis* displayed a similar survival in recent and long-abandoned pastures, but higher mortality close to the forest border.

Mexico: Central and Northern Highlands of Chiapas (Sites 22–33)

Functional groups of native tree species

Matching the tolerance of native tree species with environmental gradients that operate at the microsite level is required for successful forest restoration (Ramírez-Marcial *et al.*, 2005). Conditions occurring in restoration sites represent environmental filters that define the assembly rules of a plant community (Temperton *et al.*, 2004). Forest restoration should be based on the grouping of sets of species into functional groups whose life history attributes and population dynamics are sufficiently consistent to guide restoration actions at the plot, landscape and regional spatial scales in high diversity areas. Therefore, we have studied the main germination requirements of a large number of species (140 taxa; Ramírez-Marcial *et al.*, 2003, 2005) while producing seedlings to be used in field experiments on plantation enrichment. Some of the species studied have been classified as endangered taxa in national or international lists (Oldfield *et al.*, 1998; SEMARNAT, 2002). The tolerance to partial shade (or intolerance to open conditions) of more than 40 species has been evaluated under common nursery conditions; boxes covered with black net mesh of different openings allowing variable light incidence on the experimental plants have been used (Fig. 15.5A).

Nursery experiments on seedling response to light and water gradients

Understanding the responses of key species to environmental gradients is a crucial piece of knowledge to model and guide practices aimed at restoration of forest communities. We conducted a nursery experiment to elucidate the specific responses of seedlings of three *Pinus* spp., three *Quercus* spp. and six other understorey broadleaved tree species in a common garden: *Alnus acuminata*, *Cornus disciflora*, *Garrya laurifolia*, *Olmediella betschleriana*, *Prunus lundelliana* and *Styrax magnus*. Selection of species was based on our advances in a classification scheme of tree seedling functional groups, which considers attributes pertaining to their regeneration niche as well as to availability of seeds. The experiment started in March 2003 and included three conditions (25, 75 and 100%) of photosynthetic active radiation (PAR) and three soil moisture levels: field capacity (24%), intermediate (18%) and permanent wilting point (13%). A number of 12 replicates (20 for conifers and oaks) for each treatment combination and species were established. A total of 2064 seedlings were planted in independent plots within a common garden of $c.500\,m^2$ located at the ECOSUR facilities in San Cristóbal de Las Casas, Chiapas. The experiment ended at the start of the rainy season (end of May 2003), but some lower levels of direct sunlight (8, 15 and 25) were assessed with *Pinus* spp. and *Quercus* spp. in March–May 2004. We measured seedling survival every 2 weeks, and stem height, basal stem diameter, number of leaves, number of recently emerged leaves and leaf size of the three largest leaves. At the beginning of the rainy season, we harvested four out of ten seedlings to analyse patterns of resource allocation to different plant organs. We left six seedlings in the nursery to provide information on long-term responses to radiation (water cannot be controlled during the rainy season). Seedlings of five out of six species (but not *A. acuminata*) subjected to drier and more open conditions had higher mortality than those with heavier shade and wetter soil. Stem height, basal diameter and number of leaves were affected by shade intensity. Light conditions had the highest effect on the distribution of dry biomass in all tree species.

Underground herbivory and seedling establishment

Establishment of enrichment plantings may be affected by herbivores and root feeders. Root damage by larvae of *Phyllophaga* spp. (Coleptera: Melolonthidae) has been observed to affect seedling survival and establishment. We evaluated below-ground herbivory by two *Phyllophaga* species (*P. obsoleta* and *P. tumulosa*) on seedlings of ten native tree species (*Arbutus xalapensis*, *Litsea glaucescens*, *Myrica cerifera*, *Nyssa sylvatica*, *Persea americana*, *Quercus crassifolia*, *Quercus skutchii*, *S. magnus*, *Synardisia venosa* and *Ternstroemia lineata* ssp. *chalicophyla*). A total of 550 plants were included in the experiment and 300 seedlings were inoculated with one larva of each *Phyllophaga* species. Plants were maintained under nursery conditions for two months. Plants were harvested and oven-dried to obtain biomass of aerial and below-ground plant organs. The results indicate that herbivory of roots was significantly different for eight of the ten studied species (except *P. americana* and *S. venosa*) and damage intensity by *P. obsoleta* was higher in five tree species.

Restoration of forest edges (Sites 22 and 30)
Forest clearing in Chiapas is mostly related to establishment of slash-and-burn *milpa* agriculture (maize–beans–squash). The system may last for 2–4 years, but the use of fertilizer and herbicides may allow for permanent agriculture (González-Espinosa *et al.*, 1991, 2006; García-Barrios and González-Espinosa, 2004). Secondary forests usually develop with a variable dominance of *Pinus* spp. due to selective logging of *Quercus* spp. and other broadleaved species that are preferentially used for firewood; on the other hand, *Pinus* spp. are allowed to grow until they attain adequate sizes for timber extraction and can reproduce several times. Forest restoration opportunities arise when fallow fields, pastures and early secondary forests are left for succession to progress. In 1998 we started a study with experimental clearings (ten plots, 10 m × 10 m each; Table 15.2) at the border of forests with variable dominance by *Pinus* spp., subsequently followed by two agricultural cycles, fallow field and enrichment of shrublands. After 54 months of transplanting the saplings, the nine broadleaf tree species that were introduced (mostly old-growth and intermediate successional species; Table 15.2) show an average survival of 73% (590 alive plants out of 810). The greatest relative growth rate in height and diameter has been observed in *Arbutus*, *Clethra*, *Cornus* and *Quercus laurina*. These preliminary results indicate that enrichment of forest edges in a forested landscape does not seem to require a previous facilitation stage with light-demanding species.

Restoration essays in a variety of field conditions (Sites 23–29 & 31–33)
The central and northern Highlands of Chiapas include a wide variety of environmental conditions and the distribution of many native tree species samples these conditions extensively. To probe the involved gradients we have been keen to take advantage of offerings from interested groups to establish

Fig. 15.5. Relationship between relative growth rates in stem height (RGR_{height}) under partial shade (25% of direct light) and at full direct light in open areas for seedlings of 42 native tree species of the Highlands of Chiapas (Mexico) under nursery or common garden conditions (A), and for 24 tree species under field conditions (B). Abigua, *Abies guatemalensis*; Acapen, *Acacia pennatula*; Alnacu, *Alnus acuminata* ssp. *arguta*; Arbxal, *Arbutus xalapensis*; Budcor, *Buddleja cordata*; Chipen, *Chiranthodendron pentadactylon*; Clepac, *Clethra pachecoana*; Clethe, *Cleyera theaeoides*; Cordis, *Cornus disciflora*; Garlau, *Garrya laurifolia*; Ilevom, *Ilex vomitoria*; Liqsty, *Liquidambar styraciflua*; Magsha, *Magnolia sharpii*; Myrcer, *Myrica cerifera*; Olmbet, *Olmediella betschleriana*; Orexal, *Oreopanax xalapensis*; Perame, *Persea americana*; Pinaya, *Pinus ayacahuite*; Pinpse, *Pinus pseudostrobus* ssp. *apulcensis*; Pintec, *Pinus tecunumanii*; Plamex, *Platanus mexicana*; Prulun, *Prunus lundelliana*; Prurha, *Prunus rhamnoides*; Pruser, *Prunus serotina* ssp. *capuli*; Psygal, *Psychotria galeottiana*; Queaca, *Quercus acatenangensis*; Quecan, *Quercus candicans*; Quecra, *Quercus crassifolia*; Quecri, *Quercus crispipilis*; Quelau, *Quercus laurina*; Querug, *Quercus rugosa*; Quesap, *Quercus sapotifolia*; Queseg, *Quercus segoviensis*; Quesku, *Quercus skutchii*; Quesp., *Quercus* sp.1; Ranacu, *Randia aculeata*; Rapjue, *Rapanea juergensenii*; Rapmyr, *Rapanea myricoides*; Rhacap, *Rhamnus capraeifolia* var. *grandifolia*; Rhasha, *Rhamnus sharpii*; Simlim, *Symplocos limoncillo*; Stymag, *Styrax magnus*; Synven, *Synardisia venosa*; Terlin, *Ternstroemia lineata* ssp. *chalicophila*; Terooc, *Ternstroemia oocarpa*; and Zanmel, *Zanthoxylum melanostictum*.
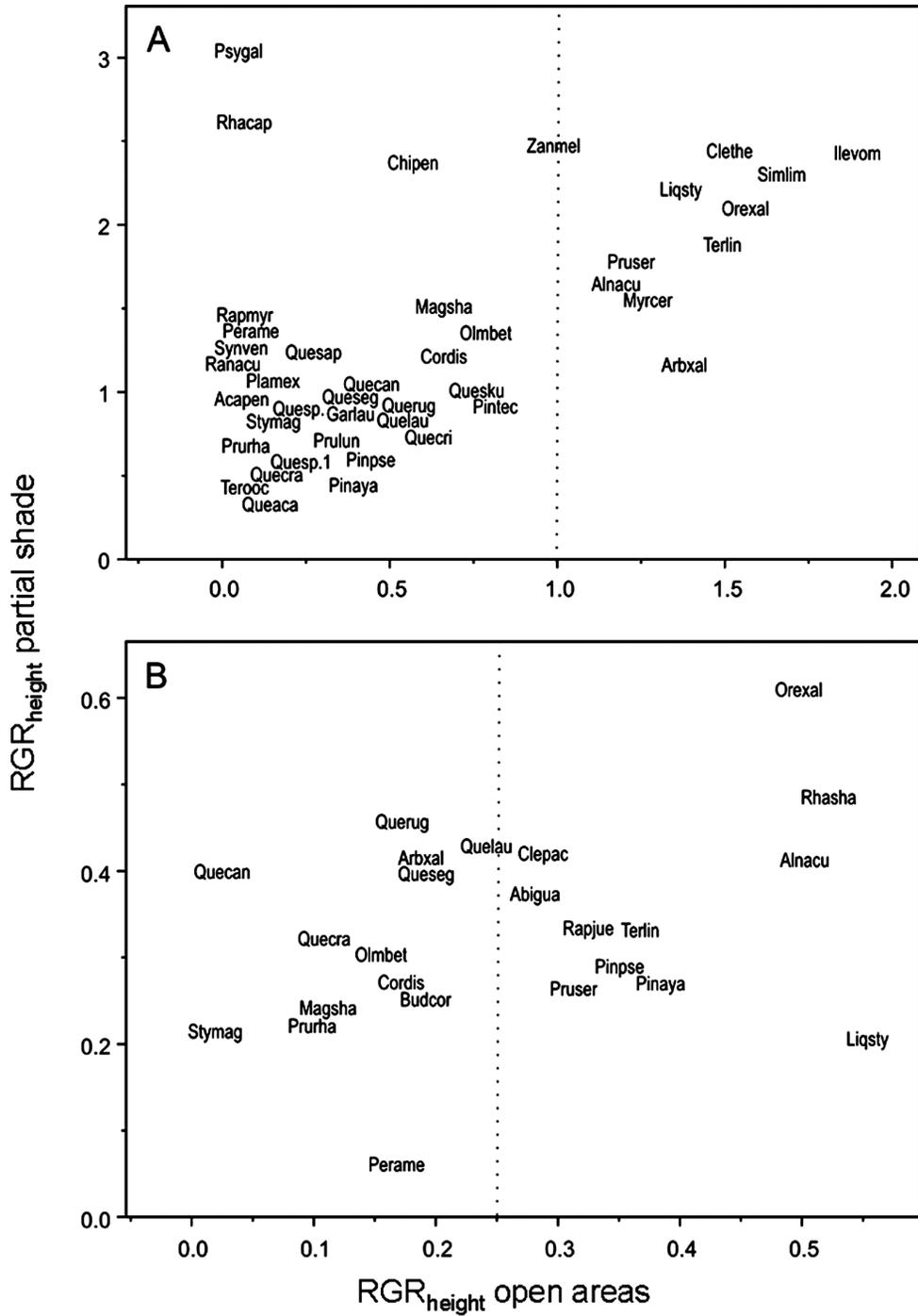
restoration essays in their lands. Therefore, a number of restoration plantations have been established and monitored (survival and growth of stem height and basal diameter) using a set of 60 species including conifers, *Quercus* spp. and other broadleaved species that can be considered as early, intermediate or late successional (Sites 26–33 in Table 15.2). The essays have been started at different times (mostly 1–3 years ago, and one study has been monitored for 15 years; Quintana-Ascencio *et al.*, 2004). Although the species were introduced in sites with different disturbance regimes, it is clear that survival after 3 years may be 30–40% in open areas, but >90% under induced pine-dominated canopies. Some species can be distinguished for their growth potential under a variety of environments (e.g. *A. acuminata, Buddleja cordata, Chiranthodendron pentadactylon, Pinus* spp., *L. styraciflua, O. betschleriana*), and it is possible to propose some species groups. For example, *Oreopanax xalapensis, Rhamnus sharpii* and *A. acuminata* are easy to propagate by seed and can establish well in open areas, in early successional forests and under *Baccharis vaccinioides* shrubs (a typical nurse plant; Ramírez-Marcial *et al.*, 1996). *Pinus* spp., *Buddleja* spp., *L. styraciflua* and *Prunus serotina* ssp. *capuli* are shade-intolerant species that can establish easily in open areas; their high growth rates induce facilitation processes for late successional species that require a previous canopy such as *Magnolia sharpii, P. americana, P. lundelliana, Prunus rhamnoides, S. magnus,* and others (Fig. 15.5). A first detailed account of the invertebrate soil fauna has been obtained in the eight restoration plots established in Site 33, which were subjected to severe fire disturbance in 1998. Abundance and diversity of the soil fauna showed marked seasonality and it includes 187 morphological species belonging to 58 families and 20 zoological orders within six classes and three phyla.

Interactions between tree seedlings and herbaceous cover (Site 30)

Forest restoration in abandoned pastures could be accelerated or arrested if tree seedling establishment is affected by competition from the surrounding herbaceous cover. Seedlings of ten native tree species (Site 30, Table 15.2) were used in experiments. In July 2003 we established 21 experimental plots (10 m × 10 m) in three grassland and four shrubland sites. Each plot included 6–10 seedlings of each species (a total of 1656 plants). In each grassland or shrubland, one plot served as control, a second one was subjected to a treatment of aerial herb removal (clipping herbs within a radius of 30 cm around each seedling), and a third plot was subjected to total herb removal (both above and underground tissues killed with herbicide application). After 22 months, the preliminary results suggest that grasses may have different competitive effects on seedlings, both above and below ground, in grasslands and shrublands.

Discussion

This integrated and synoptic report pinpoints some valuable experiences that can be considered as lessons learned, and can contribute to the development of best practice in forest restoration in our study sites and other similar areas. The large range of environmental conditions included in these studies

is matched by a wide array of socio-economic factors. Their joint consideration may lead to broad guidelines, criteria and indicators for ecological restoration that may represent many of the conditions prevailing in other developing regions. Current pitfalls can be identified and used to define a minimum set of elements to be considered in a protocol for a more widespread assessment of restoration experiences, both scientific and practical.

Ecological issues and forest restoration

Forest restoration aims to reproduce and enhance ecological processes that drive community development through time. Ecological models incorporating general principles that drive the organization of ecosystem diversity during succession are particularly relevant in this context (Bradshaw, 1987; Ramírez-Marcial *et al.*, 2005; Ruiz-Jaén and Aide, 2005). So far most of our studies have concentrated on assessment of plant performance (mostly at the seedling stage, rarely with saplings) in response to either one or many variables. As an example of this latter case we can mention treatments with and without grasses (or moss), which most probably trigger a number of non-specified interacting variables such as: (i) competition for nutrients, water and light; (ii) modification of temperature and humidity gradients in the immediate neighbourhood of the target plants; (iii) differential effects of the biota below ground, and so on. In the end, we may still be presented with major problems in explaining the results obtained and therefore in defining the best restoration practice for a particular site, i.e. conducting actual restoration. These experiences highlight the need for more inclusive research models about the most crucial processes involved. There is a lack of models that can be used to explore the assembly rules involved in the stratification of forest communities and shade and (or) drought tolerance along environmental gradients at landscape and regional spatial scales (Hobbs, 2002). Some promising models may be those aiming to explain broad macroecological patterns of diversity based on life history and population attributes (e.g. Huston and Smith, 1987; Storch *et al.*, 2005).

Successful forest restoration depends on the appropriate matching of environment with species tolerance. It is not coincidental that all of our research teams began with trying to understand the germination or vegetative propagation requirements of individual species or groups of species. This has been pursued in the first place to secure provision of adequate experimental material, but also to define protocols for widespread application of propagation techniques. Yet, unless several environmental variables are studied in a factorial way (e.g. light and water availability), our experiences with common garden or nursery experiments indicate that only preliminary and relative conclusions can be reached in comparison to field experiments. For example, relative growth rates of a considerable number of tree species were 4–5 times higher in the nursery than under a variety of field conditions in the Highlands of Chiapas (Ramírez-Marcial *et al.*, 2005), suggesting also the need for better experimental control in experiments under actual canopies (Fig. 15.5).

A neglected issue that may have implications for forest restoration practices results from traditional forest use patterns: low intensity but long duration human disturbance associated with selective and scattered logging of small trees, or harvesting of branches and resprouting stems (for firewood or non-commercial timber use; Vetaas, 1997; Ramírez-Marcial *et al.*, 2001, 2005; Barrón-Sevilla, 2002; Martorell and Peters, 2005). This may create environmental gradients inside the forest that do not match either those associated with disturbance patterns in old-growth stands (either forest gaps or sunflecks) or those involving widespread forest clearing (Méndez-Dewar, 2000). This little-studied aspect of forest heterogeneity may influence individual plant responses in restoration practices aimed at species enrichment of degraded stands.

Socio-economic issues and forest restoration

Until recently the strategies followed for conservation and sustainable use of forests, and also the role of forest restoration, have differed among the study regions. The South American cases exemplify a conservation strategy largely dependent on the availability of national parks and/or biological reserves for the conservation of particular species (Table 15.2) vis-à-vis native forest destruction driven by logging companies, establishment of industrial plantations with exotic species and activities of small farmers (the frontier model *sensu* Rudel and Roper, 1997). It would seem that coexistence between biodiversity and increased demand for agricultural products is being solved mostly through adoption of the model that couples land-sparing with high-yield farming (Green *et al.*, 2005).

In contrast, conditions prevailing in Chiapas point towards different avenues for development and conservation. Forest loss can be mostly explained by the so-called immiserization model (Rudel and Roper, 1997), which involves increasing populations of poor peasants who have scarce economic opportunities besides clearing additional land for agriculture. Yet this does not mean that the frontier model did not play a major role in the region in the 1970s and early 1980s, particularly in lowland areas (Montoya-Gómez *et al.*, 2003). In addition, many indigenous Mayan communities or their organizations have a strong interest in increasing their political self-determination over their relatively densely populated territories (Burguete-Cal y Mayor, 1999; Cartagena-Licona *et al.*, 2005). Conservation is not seen by these communities as an alternative viable land use if no short-term economic benefits are envisaged to support local development initiatives. Under this predominant scenario, which may continue for some decades into the future, forest restoration could play a crucial role in forest conservation and sustainable use, as it could contribute to wildlife-friendly farming in high diversity and complex forest landscapes (Bray and Merino, 2004; Holder, 2004; Bray *et al.*, 2005; Green *et al.*, 2005). Sustainable use and conservation of forested landscapes will depend, therefore, on coalescing scattered forested areas through new social contracts among communities that frequently

compete for economic opportunities, and finding new market values for traditional products provided by highly diverse mixed forests (including timber, non-timber and ecosystem services). It is in this context that calculating the current and future cost of restoration practices becomes an issue of utmost importance, and one for which unfortunately all contributing teams have so far only scanty information – if any.

Current trends suggest that, in the mid-term, forest restoration practice in the South American study regions and in southern Mexico may have a few more common elements than those they now share. On the one hand, original indigenous groups may be called upon along with other social actors to play an unprecedented role in forest planning in Chile (Lara, 2004), and local entrepreneurs may increasingly participate in financing a more intensive agriculture and social welfare in Chiapas that may allow setting aside larger forest areas for biodiversity conservation and ecosystem services (Cartagena-Licona *et al.*, 2005; Ixtacuy-López *et al.*, 2006). On the other hand, as has happened in Chile before (Armesto *et al.*, 1998), the participation of rather resourceful and well-educated social groups in the cities may become a key factor in local forest restoration efforts. In central Veracruz, a number of social groups based in the city of Xalapa have supported forest conservation actions and environmental education, including rehabilitation of evergreen cloud forest species and habitats (Pedraza and Williams-Linera, 2003; Williams-Linera *et al.*, 2003; Alvarez-Aquino *et al.*, 2004; Benítez-Badillo *et al.*, 2004; Suárez-Guerrero and Equihua-Zamora, 2005).

Academic institutions and forest restoration

Academic groups have to define their role as intermediate actors within the complex social scenario that forest restoration may imply (Lyall *et al.*, 2004; Castillo *et al.*, 2005). The wide spectrum of social conditions under which our restoration research has been conducted provides opportunities to focus on the activities of the research group once results have been validated and can be transferred to users and interest groups. In the IX and X Regions of Chile, the academic groups based at the Universidad Austral de Chile and Universidad Católica de Temuco have been able to organize an inclusive network of public and private stakeholders with an interest in *in situ* conservation. Their results in conservation biology research have provided the basis on which to conduct educational and outreach activities involving governmental and non-governmental organizations, university researchers and local people. A similar experience between academic groups and private landholders has occurred in central Veracruz. Progress in Chiapas is still some steps behind such outreach activities and widespread adoption of forest restoration practices. Yet, as in the Chilean case, in both regions of Mexico there is coincidence in the view that high-diversity native forest restoration and long-term and widespread conservation will only be attained if representatives of all involved social actors participate in what should be an ecologically defined common venture (González-Espinosa *et al.*, 2007). The

academic group may currently have much of the technological know-how to promote and carry out widespread restoration actions. Mature research groups may play a crucial role in providing strategic links for other social actors involved because of informal networks maintained by their senior members (Guimerà *et al.*, 2005). Yet, unless forest restoration achieves the sustained support of the thousands of people that live in and own the forestlands in question, its efforts will hardly surpass the stage of being a mere academic exercise and will fall short of impacting on public policy and decision-making circles.

Conclusions: Some Lessons Learned

We suggest that the following biological and socio-economic criteria could usefully be included among elements of best practice when starting a forest restoration programme, for either experimental or other purposes:

1. *To ensure that the biological material being used includes as much genetic variation as possible*. Recent studies provide evidence of the long-term reduced genetic variation that a founder population can impose on a regenerating secondary forest (Sezen *et al.*, 2005). Efforts should be made to ensure that any planting material used is well adapted to the sites where restoration is to take place.

2. *To obtain a reliable baseline estimate of the carbon content in the soil*. The global soil C pool is estimated to be 3.3 times the size of the atmospheric C pool and 4.5 times the size of the biotic sink (Lal, 2004). However, forest stands restored with different dominant species may differ in their potential root production and inputs to the soil C pool (e.g. pines lower than broadleaved native trees; Schlesinger and Lichter, 2001; Matamala *et al.*, 2003). As forest restoration is widely accepted as a viable alternative to increase C pools, its financial and social support can only benefit from being able to clearly show its potential advantage after some years.

3. *To approach the assessment of species with a gradient framework*. Species are usually distributed over a larger area than those used for restoration trials. Trees are long-lived species that may experience changing environments throughout their lifespan. Restoration predictions generated by models dealing with large spatial and temporal scales would benefit from a gradient approach to assess species responses.

4. *To consider major ecological principles and concepts*; in particular, assays designed to define the assembly rules of natural communities (e.g. plant succession, inter- and intraspecific competition, gene flow and inbreeding depression, nutrient cycling).

5. *To allow the potential users to define and take the first steps in the process of adopting results towards their application*. Forest restoration may be expensive, and potential users or landholders should be aware and ready to accept that application of their results may imply financial risks. Monitoring the effectiveness of restoration over large areas may only be possible if individuals or

community landholders participate in the process after receiving adequate training and capacity building.

6. *To be aware of novel or non-conventional statistical approaches for analysis that can help to make sense out of data obtained under very different conditions*. Not all restoration experiences will contribute to developing scientific understanding, but long-term data under a variety of conditions may support meta-analysis approaches. In many cases establishing forest restoration trials and experiments has depended on opportunities offered by potential users or groups of interest that set challenges beyond conventional experimental layouts. All contributing research teams have been keen to identify interest groups that are willing to support restoration activities; in fact, access to several of the study sites listed in Table 15.1 was negotiated with private or community landholders.

7. *To adopt an adaptive management approach that can take advantage of changing values of the land and the tree species being used*. The academic groups should take the responsibility of identifying and promoting new technologies that could be used to improve the resource base of their partners.

8. *To assess the current and future finances of alternative restoration programmes*. In order to be adopted, ecological restoration must be environmentally and economically sound.

9. *To use native tree species in forest restoration programmes, preferably in mixed plantations*. The original and traditionally managed forest ecosystems of southern and eastern Mexico include a very high diversity of tree species. On the other hand, the temperate forests of Chile and Argentina include a large number of endemics. Yet this guideline may enter into conflict with the increasing interest or need to establish plantations with exotic species in highly productive sites; this should be resolved stressing regional and long-term sustainability criteria, and not predominantly with local and short-term cost–benefit planning.

10. *To use low cost alternatives in the first place*. There are many situations where it may be preferable to allow forests to recover naturally through secondary succession. Yet this may be a slower process and may not include the complete regional pool of species if dispersal limitations prevail in some taxa. Restoration for stand enrichment may be complemented with the provision and valuation of ecosystem services, including non-conventional timber and non-timber products in order to provide a pay-off for the long-term process.

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