



Challenges in managing forest genetic resources for livelihoods

examples from
Argentina and Brazil

Forest Genetic Resources

Barbara Vinceti, Weber Amaral and Brien Meilleur



Deutsche Gesellschaft für
Technische Zusammenarbeit (GTZ) GmbH

im Auftrag des:



Bundesministerium für
wirtschaftliche Zusammenarbeit
und Entwicklung

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The International Plant Genetic Resources Institute (IPGRI) is an independent international scientific organization that seeks to advance the conservation and use of plant genetic diversity for the well-being of present and future generations. It is one of 15 Future Harvest Centres supported by the Consultative Group on International Agricultural Research (CGIAR), an association of public and private members who support efforts to mobilize cutting-edge science to reduce hunger and poverty, improve human nutrition and health, and protect the environment. IPGRI has its headquarters in Maccarese, near Rome, Italy, with offices in more than 20 other countries worldwide. The Institute operates through three programmes: (i) the Plant Genetic Resources Programme; (ii) the CGIAR Genetic Resources Support Programme; and (iii) the International Network for the Improvement of Banana and Plantain (INIBAP).

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Citation:

Vinceti, B., W. Amaral and B. Meilleur, editors. 2004. Challenges in managing forest genetic resource for livelihoods: examples from Argentina and Brazil. International Plant Genetic Resources Institute, Rome, Italy.

ISBN 92-9043-667-0

IPGRI

Via dei Tre Denari 472/a

00057 Maccarese

Rome, Italy

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To Dr Abdou Salam Ouédraogo (1957–2000)

This book is dedicated to the fond memory of our late colleague Dr Abdou Salam Ouédraogo (1957–2000), who died in the Kenya Airways crash on the night of Sunday 30 January 2000.

A citizen of Burkina Faso, Abdou was well known throughout Africa and the scientific community as a distinguished scientist and leader in his field. In 1995 he was awarded a PhD in conservation biology by the University of Wageningen, the Netherlands. He was the founding director of the Forest Tree Seed Centre in Ouagadougou and coordinator of the regional forest genetic resources programme at the Food and Agriculture Organization/Permanent Interstate Committee for Drought Control in the Sahel (FAO/CILSS). During his tragically curtailed career, he was an active member of several panels and committees for international organizations. He served as a member of the FAO Panel of Experts on forest genetic resources, representing Africa. He was part of the International Tropical Timber Organization (ITTO) Panel of Experts advising on forest genetic resources while also a member of the programme committees of the World Agroforestry Centre (ICRAF) and the forestry department of the Centre de coopération internationale en recherche agronomique pour le développement (CIRAD–Forêt). He fulfilled leadership roles in key conservation bodies as the deputy leader of the Research Group on Biodiversity (for Africa) of the International Union of Forestry Research Organization (IUFRO) and as chair of the Species Survival Commission African Tree Specialist Group, part of the World Conservation Union (IUCN). He was a member of external review panels for key international forestry organizations such as the Forest Seed Centre of the Danish International Development Agency (DANIDA) in 1996, and IUFRO and CIRAD–Forêt (both in 1999).

Abdou joined the International Plant Genetic Resources Institute (IPGRI) in 1993 as Senior Scientist, Forest Genetic Resources, leading IPGRI's global project on forest genetic resources. These years of 'hands-on' experience gave him a thorough understanding of the complex realities and needs of the developing world in the area of plant genetic resources, particularly in sub-Saharan Africa. It was in this capacity that he and others conceived the project 'Conservation, management and sustainable use of forest genetic resources with reference to Brazil and Argentina', presented in this book. He also gained an excellent knowledge of the *modus operandi* of international organizations, including the centres of the Consultative Group on International Agricultural Research (CGIAR), the FAO, IUFRO, CIRAD–Forêt and DANIDA, in addition to experience in leadership and management. In October 1999 he was promoted to the position of Regional Director for sub-Saharan Africa based at IPGRI's office in Nairobi, Kenya. During the short time that he was in this position, he demonstrated unique vision, leadership, an engaging personality and good humour.

We will remember him particularly for his great ability to work with people and instil team spirit, and for his vitality and his positive attitude towards life, even when faced with enormous challenges. He was a unique individual, loved by everyone who knew him. His life was a continuous endeavour to make the world a better place.

Acknowledgments

The German Federal Ministry for Economic Cooperation and Development (Bundesministerium für wirtschaftliche Zusammenarbeit und Entwicklung – BMZ), through the German Technical Cooperation (Gesellschaft für Technische Zusammenarbeit – GTZ), generously provided financial and technical support for the research activities presented in this book and made its publication possible.

The International Plant Genetic Resources Institute (IPGRI) gratefully acknowledges the experts who contributed so much to the field research, and to preparing this book's 13 chapters. We also wish to express our appreciation to project coordinators Dr Leonardo Gallo (Instituto Nacional de Tecnología Agropecuaria [INTA], Argentina), Prof. Paulo Kageyama (Escola Superior de Agricultura, 'Luiz de Queiroz', Universidade de São Paulo, Brazil [ESALQ-USP]) and Dr Antonio Higa (Universidade Federal do Paraná [UFPR], Brazil), and to the project participants who guided initial phases of the project, especially Prof. Dr Hermann Ellenberg (Federal Research Centre for Forestry and Forest Products [BFH], Institute of World Forestry, Germany), Prof. Dr Richard Stephan (BFH, Institute for Forest Genetics and Forest Tree Breeding, Germany), and Dr Rudi A. Seitz (UFPR, Brazil).

IPGRI particularly wishes to thank the local people, communities and organizations participating in the project, and especially the Mapuche indigenous communities of Neuquén Province (Northern Patagonia, Argentina), the Rubber Tappers Council of Rio Branco (Acre, Brazil), the Pontal Landless Movement (São Paulo, Brazil), and the Antônio's Community in Iraty (Paraná, Brazil).

I also wish to express my personal gratitude to the experts who added so much to the scientific discussions at the various project workshops and to IPGRI staff members who developed the research proposal. Special thanks are offered to the late Dr Abdou Salam Ouédraogo (Senior Scientist, Forest Genetic Resources), Dr Leonardo Petri (Associate Scientist on Forest Genetic Resources, supported by the Italian Ministry of Foreign Affairs), Jean Marc Boffa (IPGRI's consultant on Forest Genetic Resources, supported by the Dutch Ministry of Foreign Affairs), and to other IPGRI scientists who contributed to the project's implementation with valuable ideas and time – particularly Dr Jan Engels (Director, Genetic Resources, Science and Technology Group), Dr Pablo Eyzaguirre (Senior Scientist, Anthropology and Socioeconomics), Paulo van Breugel (Associate Expert, Forest Genetic Resources), and Walter de Boef (ETC consultant to IPGRI) who facilitated most project meetings from the inception of the initiative.

Finally, special thanks go to Barbara Vinceti (Associate Scientist, Forest Genetic Resources, supported by the Italian Ministry of Foreign Affairs), Brien Meilleur (Unité mixte de recherche 5145: Eco-anthropologie et ethnobiologie, CNRS-Paris) and Michelle Grayson of Green Ink for their editorial work, Patrizia Tazza for artwork design and Paul Philpot of Green Ink for design and page layout.

Weber Amaral

(Global Coordinator of IPGRI's Programme on Forest Genetic Resources)

Foreword

Global concerns for the fate of tropical forests continue to mount despite the fact that “progress in implementing sustainable forest management around the world has been steady and encouraging” (FAO 2003)¹. “The estimated net annual change in forest area worldwide from 1990 to 2000 was –9.4 million ha (this figure represents the difference between the estimated annual rate of deforestation of 14.6 million ha and the estimated annual rate of forest area increase of 5.2 million ha)” (FAO 2001)². In coming years, population increases combined with growing per capita consumption will continue to result in agricultural expansion into new lands, mostly through deforestation. Preliminary findings of a study by the Food and Agriculture Organization (FAO) indicate that agricultural land is expanding in about 70% of countries, and in two-thirds of these countries forest area is decreasing (FAO 2003)¹.

A large number of species found in habitats that are rapidly disappearing require conservation measures. However, our limited knowledge of the impacts of deforestation, uncontrolled exploitation and other threats to genetic diversity in tropical forests is a problem that restricts the capacity of regional, national and international institutions to plan and implement appropriate actions. There are many major challenges to developing strategies for the conservation and sustainable use of forest ecosystems and their tree species. These include the difficulties of setting priorities for intervention; scaling up research findings from local dimensions to larger scales and feeding these results into regional action plans; generating best practice for ecosystem and species management based on work limited to model species; raising awareness within national programmes in order to ensure that recommendations and guidelines for conservation and sustainable use of forests are adopted into policy and practice; and, very importantly, engaging local communities in conservation actions, making sure they can benefit directly from the sustainable management and use of forest resources.

It is important to note that past efforts at conservation, through establishing protected areas, national parks and reserves, have not given sufficient consideration to the distribution of species and their infraspecific genetic variation, both of which are central to ensuring the maintenance of adaptive capacity and production potential to meet present and future needs. Thus, research is needed at ecosystem, species and infraspecific levels in order to support the development of complementary conservation strategies and effective policies that reconcile the needs and interests of local communities and governments.

The Convention on Biological Diversity provides an overall framework for the protection of forest ecosystems and their tree species. Countries that ratified the Convention are required to assess and monitor their biological resources and to develop effective strategies for conserving them. These strategies and methods for the conservation and sustainable use of tropical forest trees should ensure that forest ecosystems make an increased contribution to the livelihoods of local communities and national economies as well as securing the genetic diversity of target species for the future.

Discussions between German partners and the International Plant Genetic Resources Institute (IPGRI) during a workshop in 1995 on ‘*In situ* conservation of plant genetic resources for food and agriculture in developing countries’ resulted in a decision that efforts related to the conservation, genetic management and sustainable use of forest genetic resources should receive high priority. Consequently, in close collaboration with two German forestry research institutes, with identified partners in Brazil and Argentina, IPGRI initiated a consultation process that ultimately resulted in the German Federal Ministry for Economic Cooperation and Development (BMZ) funding this project, entitled ‘Conservation, management and sustainable use of forest genetic resources with reference to Brazil and Argentina’.

¹ FAO, 2003. State of the world's forests, Food and Agriculture Organization of the United Nations, Rome, Italy

² FAO, 2001. State of the world's forests, Food and Agriculture Organization of the United Nations, Rome, Italy

This book presents the project activities that were undertaken within a framework that integrated socioeconomic, policy, population genetic and ecological features of four different forest ecosystems spread across South America, through a participatory and multidisciplinary research approach. Project implementation was a true learning event for all the parties involved. It resulted in unexpected problems, unforeseen delays, communications difficulties, and coordination and implementation challenges that turned the project into a pioneering experience.

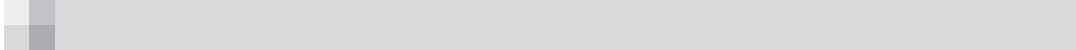
In view of the above, we all felt it was important to share the experiences of the project partners with the broader community so that others could learn from the mistakes that were made as well as from the positive outcomes. I would thus like to summarize a few of the key learning experiences, more details of which are found in subsequent chapters of this publication. First and foremost is the importance of involving key stakeholders as equal partners, from the initial stages of project planning through to the interpretation of results. It was felt that no compromises should be made and that the project proposal should adequately reflect this in the activities and budgets that correspond to this critical phase. A second, related point is the need to allow sufficient time for individual research partners to engage rural community stakeholders in the implementation of the activities that they had agreed to undertake. This required flexibility in project implementation, sometimes more than current practices allow. Thirdly, as the approaches need to be truly 'bottom up' in order to allow and ensure the required commitment and true participation of all stakeholders, it was necessary to facilitate this essential participation through adequate budgetary arrangements. It is not sufficient to assume that stakeholders have the means to participate 'at their own expense'.

On behalf of IPGRI, I would like to use this opportunity to express my whole-hearted thanks to the Brazilian and Argentinean scholars, institutions and local communities, and NGOs who contributed in so many ways to this project; to the German researchers for their partnership; to BMZ as the donor; and to the German Technical Cooperation (GTZ) as the implementing agency. Their collective patience and understanding for the delays and changes needed to bring this project to fruition are enormously appreciated. I would also like to thank my IPGRI colleagues who invested so much time and effort to coordinate the implementation of this project. And finally, to my friend and colleague Abdou Salam Ouédraogo, who unfortunately could not witness how significantly 'his' project has contributed to our common goal of conserving and utilizing the dwindling forest genetic resources of this world, I express my most sincere gratitude. I hope that this publication will further contribute to this aim.

Jan Engels
Director, Genetic Resources Science and Technology Group
IPGRI, Rome, Italy
Rome, December 2004

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Introduction

Land is a valuable, but limited resource. Hence there is great pressure to change the way in which land is used, which is now leading to steady losses and increased isolation of habitat remnants throughout the world. Recognition of the urgency and the extent of forest loss has spurred an increasing number of studies on forest conservation and sustainable use. Most natural-resource planners recognize genetic diversity and its underlying processes as essential components of ecosystem and species stability, adaptability and conservation, but rarely is there explicit provision for the conservation of genetic diversity in management planning and decision-making.

Conservation of genetic diversity is essential for many reasons. Among these are the adaptation of populations to changing environments, the risks to short-term seed and population viability from inbreeding depression, and the need to maintain genetic resources for possible future use. Therefore, goals for the conservation of genetic diversity must include maintaining the variation that affects the fitness of individuals, provides for adaptation to future environmental change, and permits ongoing genetic processes such as gene flow and natural selection to occur while genetic drift is minimized (Namkoong 1993)¹.

Given the pervasiveness of the processes of deforestation, landscape fragmentation, ecosystem simplification and associated species loss, there is an urgency to the task of integrating human population needs with the preservation of essential ecosystem processes. Some communities of forest and/or forest margin dwellers (indigenous or non-indigenous) still depend largely on forest resources for subsistence and income generation. However, increasingly, these communities are becoming disconnected from forest resources, and their economic strategies have become a less important element of community forestry-based conservation. There is therefore a need to first examine the nexus between the interests of resource users and the objectives of biodiversity conservation, and second, to analyze how current patterns of use affect the long-term maintenance of forest genetic resources (FGRs).

This book presents research findings from case studies that have explored relationships between current patterns of forest use and their effects on FGRs in four forest ecosystems.

The volume is divided into three sections. Chapters in the first section address themes that permeate the project case studies: the main threats to FGRs, the relationship between forest conditions and community-based forest management, the magnitude of extraction of nonwood forest products (NWFPs) and their current economic importance, and modelling approaches that support the management of forests and particularly FGRs.

The seven chapters in the second section outline the structure of the project funded by the German Federal Ministry for Economic Cooperation and Development (BMZ) and the results that were generated from the four research sites during the four years of project activities. These include descriptions of the first meetings with local stakeholders and research partners, identification of the species investigated, discussion of the methodologies adopted, and the use of the data obtained in order to implement models that portray the relationships among social dynamics, natural resource management and forest genetic processes.

There are two chapters in the third and final section. The first of these examines the degree of success that resulted from the adoption of a participatory approach during the research phase, and the difficulties of implementing a fruitful connection between the research disciplines (socioeconomics, species ecology, reproductive biology and genetics). It also discusses the lessons learned from project experience and provides general considerations on how to achieve better stakeholder participation and true interdisciplinary research. The second chapter in this section presents examples of practical applications for FGR research in the management of forest resources.

¹ Namkoong, G. 1993. A gene conservation plan for loblolly pine. *Can. J. For. Res.* 27:433–437

The BMZ-funded project illustrated in this book contains core elements of the FGR research that is undertaken around the world within the International Plant Genetic Resources Institute (IPGRI) programme. This programme is oriented towards the formulation of better FGR management practices that will simultaneously maintain ecosystem health and improve local community livelihoods while focusing on species that are overexploited, threatened or play particularly important roles in local economies. The dissemination of research findings from the BMZ-funded project via this publication is also an important part of IPGRI's FGR programme of increasing awareness about the need for FGR conservation and sustainable use. This is so that stable, diverse and healthy forest ecosystems will be able to provide a wide range of products and services to local as well as global beneficiaries for many years to come.

The book's 13 chapters are briefly introduced below.

Chapter 1 considers various impacts of human interventions on FGRs, including the biological consequences of major threats to forest ecosystems and associated tree species. A range of examples illustrates the effects of various types of disturbance on genetic diversity and on gene flow in species with different characteristics and within different ecosystems. Criteria and indicators for monitoring the conservation of genetic diversity are examined along with several modelled approaches to FGR conservation and use. The chapter then explores challenges to the conservation and sustainable use of FGRs from the vantage of species biology, conservation priority-setting and the allocation of funds.

Chapter 2 analyzes the principal factors that have influenced the success or failure of community-managed forest resources in a range of South American contexts. It reviews and summarizes the findings of a growing body of literature that deals with this issue and then examines these within three South American case studies. A descriptive analysis of the International Forestry Resources and Institutions (IFRI) methods in documenting principal forest characteristics in South America is presented in order to highlight the opportunities and challenges for local forest-user groups.

Chapter 3 illustrates how NWFPs are crucial resources for livelihoods in many parts of the world. This chapter addresses issues related to the potential of NWFPs for income generation at local, national and global levels. It examines how using and trading NWFPs affects the sustainability of different extraction regimes and livelihood strategies, with a focus on South America and particularly Brazil. Challenges in defining exactly what NWFPs are and in monitoring their sustainable use are highlighted, with recommendations made for further research.

Chapter 4 demonstrates how models can help us understand biological processes and identify management options. It describes the growing use of models in the study and management of natural forest ecosystems, forest plantations and forests disturbed by humans, and presents the range of their applications (from predicting forest growth and yield to estimating the effects of natural and human-influenced disturbances on forest ecosystems). The chapter then examines in detail several models adopted in FGR studies to simulate forest genetic dynamics in order to predict the effects of human interventions on forest ecosystems.

Chapter 5 introduces FGR and associated conservation issues in South America. It presents the structure and objectives of the BMZ-funded project in conservation, management and sustainable use of FGRs in Brazil and Argentina, undertaken under the auspices of IPGRI.

Chapter 6 presents the first BMZ-funded project case study that investigated threats to the genetic resources of *Araucaria araucana* in Argentina. Genetic processes were examined in *A. araucana* forests that were differently affected by human activities and along an environmental gradient, and then analyzed in relation to biological dynamics and socioeconomic conditions. Suggestions are made on how to incorporate research results into guidelines for the sound management of *A. araucana* FGRs.

Chapter 7 assesses the conservation status of another araucaria species, *A. angustifolia*, in the State of Paraná, Brazil. This BMZ-funded study investigated the repercussions of FGR access and use policy on the conservation status of mixed

A. angustifolia forests. This species dominates one of the most important naturally occurring biomes in south and southeastern Brazil that has been subjected to dramatic overexploitation.

Chapter 8 provides a brief overview of the evolution of land use and land tenure regimes in the Brazilian Amazon, illustrated by examples from Acre State. The genetic and ecological characteristics of four NWFP species are assessed in two types of rural settlements: Settlement Projects (Projetos de Assentamento or PAs) and Extractive Settlement Projects (Projetos de Assentamento Extrativista or PAEs). These two types of settlement were chosen because they were characterized by different tenure and land-use regimes that it was believed would differentially affect natural resources. The chapter reviews the settlement types and how they were created, and then makes recommendations for improved forest management practices based on the research findings.

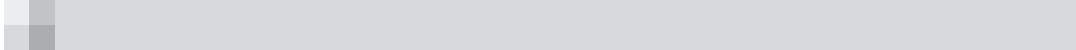
Chapter 9 describes the current situation of the FGRs of selected species, chosen in consultation with local stakeholders and found within heavily degraded fragments of the Atlantic forest in the eastern corner of the State of São Paulo (Pontal), Brazil. Key findings and their interpretation are preceded by a description of the events that led to the reduction of the forest in the Pontal region, and to the degradation of what was once a highly diverse semideciduous forest biome.

Chapter 10 provides the results of a modelling exercise undertaken at one of the *A. araucana* (pehuén or araucaria) forest research sites in Argentina. The objective was to compare the dynamic behaviours of araucaria forests subjected to different use regimes and ultimately to identify the factors that influence forest ecosystem genetic processes. The goal in applying the model was to simulate, determine and then to monitor sustainable levels of FGR use for both management and conservation purposes.

Chapter 11 presents the results of further modelling of FGRs in the vulnerable *A. araucana* forest ecosystems of Argentina. Modelling was carried out to determine how genetic diversity was spatially distributed throughout the species' range in that country. Cline theory was applied in a two-dimensional landscape analysis of selection pressures, gene flow and species distribution patterns to predict areas of high genetic diversity within the *A. araucana* range. The study provides a promising first step towards developing predictive tools for genetic conservation, both in *A. araucana* and in other species, as well as a novel means of assessing spatial evolutionary processes.

Chapter 12 assesses the performance of the participatory methods used in the BMZ-funded project and across a range of social and environmental contexts in Brazil and Argentina. An analytical framework was employed to evaluate the degree to which partnerships were developed between researchers and local people in forest resource management and policy development. Lessons learned from the BMZ-funded project are outlined in the latter part of the chapter, emphasizing both the need for capacity building among researchers and for longer time frames to implement participatory research based on effective partnerships.

Chapter 13 illustrates the practical implications of FGR research in addressing forest management issues. A portfolio of case studies is drawn from the recently concluded BMZ-funded South American research project and from other studies carried out by IPGRI in collaboration with research institutes and national research centres in developing countries elsewhere. The cases highlight the difficulties encountered when trying to scale-up research findings, extrapolate best practices from investigations limited to model species, or find fertile sociopolitical environments likely to adopt proposed solutions.





Part 1

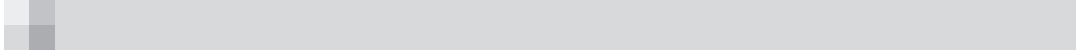
Issues Concerning Forest Genetic Diversity

**Threats to forest ecosystems and
challenges for the conservation and
sustainable use of forest genetic
resources**

**Challenges and opportunities for
communal forest management in
South America**

**Extraction of nonwood forest
products**

**Modelling the biological processes:
from genes to ecosystems**



Chapter 1

Threats to forest ecosystems and challenges for the conservation and sustainable use of forest genetic resources

D. Boshier¹ and W. Amaral²

¹ *Oxford Forestry Institute, Department of Plant Sciences, Oxford, UK*

² *International Plant Genetic Resources Institute (IPGRI), Rome, Italy*

1. Introduction

By its nature biodiversity is complex and multifaceted (e.g., it relates to ecosystems, habitats, species, populations and genes). Most natural-resource planners recognize genetic diversity and its underlying processes as essential components of ecosystem and species stability, adaptability and conservation, but rarely is there any explicit provision for the conservation of genetic diversity in management planning and decision-making. As many aspects of genetic diversity are hidden from view, its importance can easily be overlooked. Resource limitations can also result in genetic information being ignored or allocated only minor consideration in conservation or management strategies. In fact, it is still much more common for ecological, social or economic criteria, alone or in combination, to drive conservation decisions. If well-conceived goals are to be achieved in resource management and in the maintenance of long-term evolutionary adaptability of species and ecosystems, then foresters, conservationists and geneticists must establish the conditions under which genetic considerations can become integral elements of the information that is drawn upon to determine the practical conservation of species or the management of protected areas.

Conservation of genetic diversity is essential for many reasons. Among these are the adaptation of populations to changing environments, the risks to short-term seed and population viability from inbreeding depression, and the need to maintain genetic resources for possible future use. Therefore, goals for the conservation of genetic diversity must include the maintenance of variation that affects the fitness of individuals, provides for adaptation to future environmental change, and permits such ongoing genetic processes as gene flow and natural selection to occur while genetic drift is minimized (Namkoong 1993). With their unique life history traits, such as longevity, late attainment of reproductive maturity, greater opportunity for accumulation of mutations (Williams and Savolainen 1996), and their range of mating and dispersal systems, forest trees pose many challenges for genetic conservation. For instance, long life cycles produce overlapping generations, which in turn reduce effective population sizes and consequently imply greater minimal area requirements for conservation. Also, trees generally carry heavy genetic loads of deleterious recessive alleles (Williams and Savolainen 1996), such that inbreeding, and in particular self-pollination, may lead to reduced fertility and poorer regeneration, slower growth rates, limited environmental tolerance and increased susceptibility to pests or diseases (Sim 1984; Griffin 1990). The need to reduce the possibility or impact of inbreeding by maintaining genetic diversity in trees is clear, and indeed it may be critical to seed collections that are used for tree breeding programmes or for *ex situ* conservation.

This chapter considers various impacts of human disturbances on forest genetic resources (FGRs) and examines the biological consequences of major threats to forest



ecosystems and associated tree species. A range of examples illustrates the effects of different types of disturbance on genetic diversity and gene flow in species with different characteristics and within different ecosystems. Criteria and indicators for monitoring the conservation of genetic diversity are examined, along with modelled approaches to FGR conservation and use. The chapter also explores challenges to the conservation and sustainable use of FGRs from the vantage point of species biology, conservation priority-setting and the allocation of funds.

2. What are the genetic impacts of human activities on trees?

Human activities affect FGRs in many ways. Some of the activities that will be considered are conversion of forests to agricultural and other land uses, forest fragmentation, logging, use of FGRs in domestication or breeding programmes, and reforestation. All of these activities can be expected to influence genetic diversity in trees to greater or lesser degrees. For example, deforestation may lead to losses in genetic diversity, genetic bottlenecks, increased differentiation among populations and genetic isolation. Ecosystem fragmentation may alter patterns of pollen flow, leading to inbreeding or outbreeding depression and to the poor performance of seeds in plantations or habitat restoration programmes. Patterns of disturbance may also alter local environments, thereby changing selection pressures on remnant populations. The use of non-native species or nonlocal provenances of native species in reforestation may lead to hybridization with native plants, causing genetic pollution or even extirpation of native species or populations. Logging and associated management regimes may result in dysgenic selection, while over-harvesting of fruits may restrict regeneration (see Chapter 6, *Araucaria araucana*). Atmospheric pollution and climate change are more subtle disturbances, but they will nevertheless profoundly affect the adaptation of tree populations (e.g., Schmidting 1994; Mátyás 1996; Geburek 2000). Understanding the influence of human interventions on the ecology of tree populations and the relevance of these changes to the conservation of genetic diversity, and how well current conservation strategies deal with them, are important considerations in the sustainable management of FGRs.

2.1 Deforestation and fragmentation

Since the time when humans made the transition from hunting and gathering to agriculture, land-use changes have resulted in steady losses of natural habitat and increasing isolation of natural habitat remnants around the world (Henle *et al.* 2004). Given the current worldwide pervasiveness of habitat fragmentation, integrating the conservation of essential ecosystem processes with human population needs is urgently needed (Henle *et al.* 2004). Nevertheless, successful integration first requires addressing many research and management issues. This requires the development of tools for quantifying risk and the prediction of species sensitivity to fragmentation, along with the continued development of theory and methods for reassembling fragmented landscapes (Melbourne *et al.* 2004).

Fragmentation of natural ecosystems may have obvious results, such as the elimination of species, but there may also be less immediate effects on the longer-term viability of species through the modification of ecological and genetic processes within and among populations. More specifically, forest fragmentation may be seen as having three main effects:

1. reduction in the numbers of individual trees,
2. reduction of population sizes as individuals are restricted to smaller forest fragments, and
3. spatial isolation of remaining populations and individuals within nonforest land-use matrixes (Young and Boyle 2000).

The genetic consequences of fragmentation may be seen in losses of diversity at population and species levels, in interpopulation structural modifications and in increased

inbreeding. Such changes may reflect the effect of nonrandom sampling, of short-term genetic effects (genetic bottlenecks and modified gene flow), or longer-term outcomes like genetic drift. However, the occurrence and extent of such predicted changes will depend in large measure on the degree to which deforestation is random, and on whether the physical fragmentation of the forest is reflected in the segregation of the remnant forest into genetically isolated patches.

Protected areas and conservation of FGRs

In contrast to the conservation of crop genetic resources and its historical emphasis on *ex situ* approaches, FGR conservation has accentuated *in situ* methods. Nevertheless, even though national parks and forest reserves exist all around the world, few of these have been established on genetic principles. Their selection and placement have usually been determined by nongenetic criteria at the ecosystem level, with reserve design and management influenced most often by political, social and economic factors. The location of many forest reserves in hilly regions, areas of low fertility and sites of low economic value, strongly bias their value for FGR conservation. For example, the extensive removal of natural forests from sites suitable for agriculture has occurred to such an extent in some countries that many tree species are now thought to be characteristic of river courses or mountain tops, even though previously they were far more widespread. Many large tree populations and in some cases even whole forest types have completely disappeared in several parts of the world, thereby reducing the genetic configurations of some species solely to limited and highly biased representations of their former diversity. As a consequence, the conservation potential of FGRs within existing protected areas may be limited (Ledig 1988).

Which part of diversity to conserve

Neel and Cummings (2003) demonstrated that in the common absence of genetic diversity data it would be necessary to conserve 53–100% of populations to capture all the alleles of a species. The same authors argue that it would be necessary to conserve 20–64% of populations in order to represent heterozygosity reliably. However, ecogeographical variation in the distribution of tree species is generally linked to ‘locally’ common alleles (those that exceed 0.1 frequency in only one or a few populations) that make up an appreciable fraction of the genetic resources of many tree species. Despite their generally low frequency over a species’ genetic range, such alleles may be crucial to conferring adaptive potential to a species under changing environmental conditions, and they are important to effective sampling and conservation of FGRs (Brown and Hardner 2000).

Gene flow in fragmented forests

Fragmentation of forests decreases the size of tree populations and increases their spatial isolation, and these processes will influence gene flow and the genetic structure of a forest stand. Ultimately, genetic isolation of stands via a curtailment of gene flow among forest fragments will negatively affect the evolutionary viability of a population by increasing levels of inbreeding and random genetic drift (Young *et al.* 1996). However, the extent of the disruption of genetic connectivity among spatially isolated stands, and of their conservation value, is the subject of debate (e.g., Saunders *et al.* 1991; Heywood and Stuart 1992; Young *et al.* 1996). The pessimistic view that remnant trees or small forest fragments in agroecosystems are the ‘living dead’ (Janzen 1986), with little or no conservation value, is countered by data suggesting greater optimism. Substantial gene flow among isolated trees of many taxa, resulting from pollen transport by animals or wind, shows that remnant forest patches and trees are still effective in conserving genetic diversity (Hamrick 1992).

A number of studies of neotropical tree species have shown that the movement of tree pollen increased by distances of up to several kilometres following forest fragmentation (e.g., Dick 2001; White *et al.* 2002; Dick *et al.* 2003). In a study of pollen flow within continuous forest versus island populations of the self-incompatible species

Spondias mombin, the continuous (control) forest showed pollen immigration rates of 45% at >100 m distances, while in the island populations the most effective pollination (60–100%) was from pollen originating at least 80–1000 m away (Nason and Hamrick 1997). However, the more isolated islands did show reduced seed set, apparently due to a lack of effective cross-fertilization. In the self-incompatible species *Enterolobium cyclocarpum*, a dominant tree of seasonally dry forests and associated pastureland in Central America that is pollinated by bees and hawk moths, no differences were found in the outcrossing rates between trees in continuous forest (t_m 1.00; t_m = outcrossing rate based on multiple loci) and those in pastureland (t_m 0.99; Rocha and Aguilar 2001). Extensive pollen flow was found among fragments 250–500 m apart, with isolated pastureland trees experiencing more pollen donors than trees located within clumps (Apsit and Hamrick n.d.).

Nevertheless, other studies of fragmentation effects on forest tree populations suggest that another possible outcome may be the reduction of genetic diversity in small remnants through the generation of genetic bottlenecks (Young *et al.* 1993; Prober and Brown 1994). Whether a rather rapid loss of alleles through reduced population size is followed by further long-term genetic effects, such as lower heterozygosity levels or genetic drift, will depend on the extent of gene flow within and among the fragmented stands (Friedman and Adams 1985; Schnabel and Hamrick 1995; Young *et al.* 1996). Studies of *Acer saccharum* in Canada (Foré *et al.* 1992; Young *et al.* 1993; Ballal *et al.* 1994) found no signs of reduced genetic variation within remnant stands in comparison to control populations, suggesting little genetic drift in the years following fragmentation. In contrast, Prober and Brown (1994) found allelic loss and a reduction in genetic variation in small remnant stands of *Eucalyptus albens* in southeastern Australia, but only when they were separated by at least 250 m from larger populations.

Regeneration in small populations

Small populations of some broad-leaved tree species may be particularly susceptible to loss of incompatibility alleles. These alleles control mechanisms that regulate mating by imposing a 'barrier' between pollination and fertilization. This is explained as an evolutionary mechanism that prevents self-fertilization and promotes heterozygosity. Self-incompatibility is regulated by one or more loci that may have 50 or more alleles in large populations. If the same allele is present in a pollen grain and the stigma, fertilization by that pollen grain will not be successful. A lack of self-incompatibility alleles may directly threaten a population's viability through a reduction or failure in seed production. A well-documented study of regeneration in small populations focused on the threatened daisy *Rutidosis leptorhynchoides* in southeastern Australia. Regeneration was limited by the small number of incompatibility alleles that reduced the frequency of compatible crosses in small, isolated populations (Young *et al.* 2000). The small daisy populations also showed an increase in the extent of correlated mating, such that effective population sizes (i.e., the number of plants actually contributing to reproduction) and gene pools were reduced. A study of mating patterns and regeneration of the tree *Symphonia globulifera* in fragmented and continuous tropical forests in Costa Rica (Aldrich and Hamrick 1998) showed similar effects. Here, the forest fragments studied were superficially healthy, with much higher seedling densities than those in the control forest. However, 52.5% of the seedlings in the forest fragments were fathered by only two pastureland 'super adults'. Such reproductive dominance by a few trees reduces effective population size with losses of genetic diversity in subsequent generations.

Thus, for forest trees, the consequences of fragmentation are varied and complex. Studies such as those cited above show that while reductions in remnant population sizes can erode genetic variation because of bottlenecks generated at the time of fragmentation, for many tree species pollination has the potential to maintain genetic variation even over large distances under fragmentation. This picture contrasts with some traditionally held views that genetic effects such as those caused by reduced gene flow among fragments leading to losses in genetic diversity will always result from spatial isolation and population size reductions (Saunders *et al.* 1991).

Despite this more positive assessment of fragmentation, there will still always be distances between fragmented populations beyond which genetic isolation will occur, with its associated problems of viability and long-term adaptability (Young *et al.* 1996). Genetic isolation thresholds will vary among species depending on spatial structure, the presence and strength of self-incompatibility mechanisms, pollinator characteristics and availability, and the specificity of tree–pollinator relationships.

Impact of fragmentation on pollinators

Indeed, changes in pollinator assemblages in fragmented landscapes may strongly affect patterns of gene flow and reproduction in remnant tree populations. This may occur to such a degree that pollinator management (e.g., knowledge of nesting requirements or of alternative food sources for birds or insects) may be as important as managing the trees themselves. Thus, growing concerns about declines in pollinator populations in agroecosystems, and about how these declines may eventually limit tree reproduction, must be addressed (Allen-Wardell *et al.* 1998). In this regard, the many tree species with broad-range, nonspecialist pollinator assemblages are probably far less susceptible to habitat disturbance than those species with more specialized or restricted range pollinators (e.g., Dick *et al.* 2003).

Whether greater physical isolation of certain tree species results in increased self-pollination or not appears to be controlled in large measure by whether those species have self-incompatibility mechanisms. Self-compatible species that normally show some level of outcrossing, or are only weakly self-incompatible, show increased levels of inbreeding at much shorter distances of physical separation than do strongly self-incompatible species (Murawski and Hamrick 1992a). The latter species appear more likely to be characterized by a definitive physical distance beyond which seed production is substantially reduced (e.g., Ghazoul *et al.* 1998). Locally rare species are probably most vulnerable to deforestation and fragmentation. The reduction in potential partners with which to mate leads to greater self-pollination and increased inbreeding and seed failure.

While tree species with different genetic profiles, breeding systems and pollinators will react differently to forest fragmentation and isolation, evidence to date supports the conclusion that forest remnants and isolated trees on farms should be included in the design of reserves and associated corridors for conservation programmes. ‘Isolation’ of an individual tree or of a forest fragment may be more of a human perception than a biological reality, such that remnant populations and trees may actually play significant roles in maintaining genetic diversity of species.

2.2 Logging

The impact of logging on FGRs depends on many factors. As Ledig (1992) has noted, some of the more important issues are the forest type (e.g., mixed-age forests, single-age forests, monospecific forests), the portion of a species’ breeding population that is logged (e.g., selective logging or clear-fell), the seed (or other) source of subsequent regeneration (e.g., existing seed or sapling bank or postlogging fruiting), and the reproductive ecology and genetic profile of the exploited species and consequent changes to flowering, mating and fruiting patterns. Factors associated with forest typology, the methods and extent of logging, and regeneration strategies as they affect FGRs are highlighted in this section.

Logging in relation to reproductive maturity of trees and sex ratio

The genetic impact of logging on regeneration depends on the size at which trees start reproducing and on the proportion of reproductive trees left after logging. At one extreme are monocarpic species (trees that flower and fruit once in their lives, such as *Tachigali versicolor*) where logging before fruiting would have disastrous consequences for the regeneration and ultimately the survival of the species. More often, when the diameter of the trees at which logging is permitted is too small, that is, before reproductive maturity

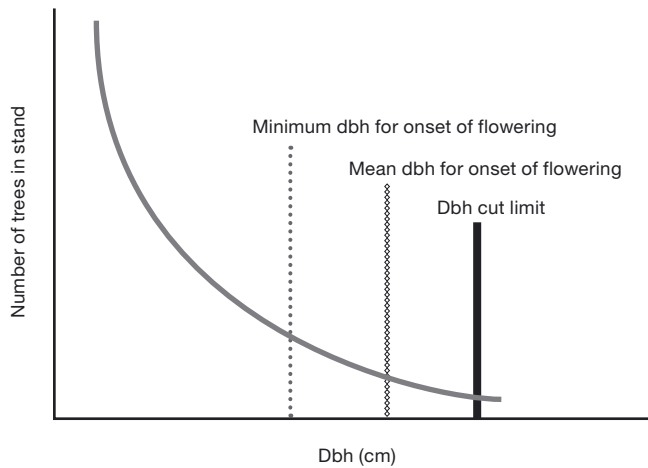


Figure 1. Relationship between size of trees at flowering and at logging. When minimum diameter at breast height (dbh) for logging is too small, a lower proportion of trees (or even none) will have reached reproductive size, and the effective population size will be reduced.

has been reached, effective population size will be reduced (Figure 1). Current data from Ghana indicate that most commercially important tree species reach reproductive size well below the 50-cm dbh (diameter at breast height) allowable cut limit (Hawthorne *et al.* 1999). However, for many tropical tree species, there is currently a dearth of information on how flowering and fruiting patterns of remaining trees are altered by logging, and on the size at which remaining individuals become reproductively mature.

Specific types of forest management may also affect reproduction by altering sex ratios. A survey of community forest reserves in the Okavango valley in Namibia found that villagers preferred logging male trees while retaining female trees in species favoured for fruit collection (Robinson 1996). Such a strategy could alter the genetic base of the species by reducing its effective population size, and might ultimately affect fruit production if pollination became a limiting factor. Species' regeneration capacities also have the potential to be influenced when fruits are harvested for food or for other nonwood forest products. Ascertaining sustainable harvest levels for fruits should be an important research priority in many parts of the world (Chapter 3 in this volume).

Species-selective logging

Logging practices in the tropics are generally species selective, that is, they remove large trees (above stipulated minimum dbh) of preferred species. Scientists will commonly make statements asserting that logging in the tropics results in dysgenic selection, that is, it will lead to losses in genetic diversity and will increase the occurrence of inbreeding within remaining trees. The extent to which logging the best-formed trees will lead to genetic erosion (dysgenic selection) is an empirical question. In order to answer this question it is first necessary to clarify what is meant by dysgenic selection: it is selection that is detrimental to the genetic quality of a population as it affects such important phenotypic traits as stem straightness or growth rate. For example, the harvesting of straight-stemmed trees prior to fruiting will normally lead to a dysgenic selection for trees of poor stem straightness if the phenotypic variation in form has a strong genetic basis (heritability h^2 is high). (Dysgenic selection is shown in Figure 2B; selection differential, S is large, where S is defined as the difference between the mean of the individuals selected to be parents and the mean of the overall population, so that S represents the average superiority of the selected parents. The change (R) in the population mean for any variable from one generation to the next is given by the equation: $R = h^2S$.)

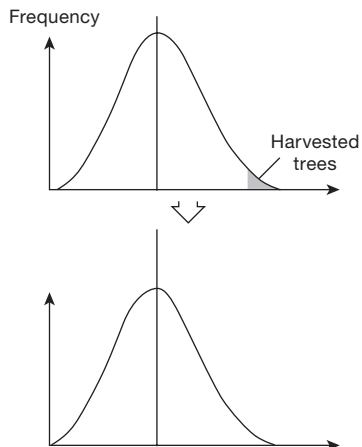
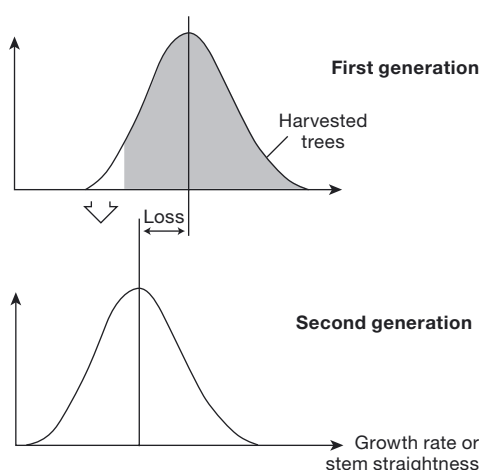
A. Harvesting exceptional trees**B. Leaving a small number of culls**

Figure 2. Impact of different logging scenarios on some traits (e.g., growth rate or stem straightness) after one generation. 2A shows how harvesting only a small proportion of trees will minimally affect the population mean, whereas in 2B, clear-fell, leaving a few poorly formed seed trees, will strongly affect the mean (e.g., of stem straightness). Actual changes will depend on the heritability of the trait of interest, as well as the selection differential. Source: After Ledig 1992.

If only a small proportion of straight mature trees are removed, the change in the population mean from first to second generation (Figure 2A) will be insignificant. In reduced-impact culling, which is now typical of approaches to sustainable logging in tropical forests, the number of trees logged in a compartment will vary from as low as 2% to as many as 50%, depending on the species. Given that diameter is usually the main criterion for determination of yield, it is unclear how much selection for stem straightness occurs, especially if selection differentials for stem straightness are low. If size mainly reflects age, rather than growth rate, there will also be little selection against growth rate. Heritability for both stem straightness and growth rate is notoriously low (e.g., <0.1) in the highly heterogeneous conditions of mixed age natural forests. Thus, it is unlikely that low extraction rates would lead to genetic erosion. Only where virtual clear-felling results in regeneration from seedlings of only a few remaining poorly formed seed trees (Figure 2B) would there be a marked deterioration in stem straightness in subsequent generations, even with low heritability of form (Ledig 1992).

Such theoretical considerations suggest instances where genetic erosion is likely to occur. The greatest potential for genetic erosion occurs in forests that have been over-logged to the point of near disappearance, leaving only a few seed trees to assure regeneration. For instance, logging for timber on St. Helena of all but a few individuals of *Trochetiopsis erythroxylon* and *T. melanoxylon* reduced both species to shrubby malformed trees (Rowe and Cronk 1995). In cases where felling or fires resulted in the removal of a high proportion of mature trees, leading to gaps in diameter classes, there may be grounds for concern about deterioration of the genetic base. But even then the situation is rarely simple. While removal of a whole generation of mature trees may adversely affect regeneration, such an action may not exert selection pressure if regeneration eventually comes from an existing cohort of poles or saplings.

Examples of logging impact on FGRs from temperate forests

There are relatively few studies of the impact of logging on genetic diversity and mating in trees, and those that have been done concentrate on northern-hemisphere conifers.

Typically, in temperate coniferous forests, clear-felling followed by regeneration predominates, but selective felling, previously more typical of high-value timber extraction from broad-leaved and tropical forests, has recently increased. Temperate forests that are subject to natural disturbances such as fire and/or wind-throw generally contain trees that are adapted and relatively resilient to disturbance (Wickneswari and Boyle 2000). Using *Pinus contorta* in Alberta, Canada, as an example, Thomas *et al.* (1999) found no significant difference in levels of genetic diversity among nonharvested stands and regenerated trees. Similarly, no significant differences in genetic variation were found in the US state of Oregon among stands of *Pseudotsuga mensiesii* managed under a shelterwood regime and those left as uncut controls (Neale 1985).

In contrast, comparison of preharvest and postharvest gene pools in old growth stands of *Pinus strobus* in Ontario, Canada, showed a loss of genetic diversity after 75% of the breeding population was harvested with seed trees left behind (Buchert *et al.* 1997). Reductions were seen in the percentage of polymorphic loci and the number of alleles, though heterozygosity levels were maintained. 'Private' alleles, those alleles unique to each stand, were especially vulnerable to harvest-induced elimination from the gene pool, with losses of 25% or more. Private allele losses of this magnitude may be common when these types of temperate coniferous forests are harvested at such intensities, raising concerns for the integrity of locally adapted gene pools after harvesting. Nonetheless, it is possible that genetic integrity may ultimately be restored during regeneration through the extensive gene flow typical of such conifers.

Effects of regeneration practices on FGRs

The effects of regeneration practices, like aerial resowing or use of seed trees following site preparation by burning or mechanical disturbance, were studied with DNA markers in southeastern Australia (Glaubitz *et al.* 2003a and b) using the commercial species *Eucalyptus sieberi* (silvertop ash) and an uncommon noncommercial species *E. considaniana*. Using several genetic diversity measures (allelic richness, number of alleles, heterozygosity) *Eucalyptus sieberi* showed no significant differences (Glaubitz *et al.* 2003a). Nor was evidence found for recent bottlenecks. However, a dendrogram of the relationships among the sampled populations suggested that the seed-tree system might promote genetic drift, while aerial resowing after clear-fells with the same seed lot would lead to genetic homogenization. The apparent genetic resilience of *E. sieberi* in native forest regeneration was attributed to its local abundance and to the favourable characteristics of its reproductive biology, where seed is held on its branches for several years and good regeneration occurs from seed already on site. The presence of *E. considaniana* typically fell from 13% prior to logging to 5% in regenerated coupes. Significant reductions were also seen in *E. considaniana* genetic diversity (allelic richness, number of alleles, heterozygosity) in seed tree coupes, but not in clear-fell areas (Glaubitz *et al.* 2003b), while there was no evidence of increases in inbreeding under either management system. Eldridge *et al.* (1993) found that family structure in native eucalypt forests leads to a certain amount of inbreeding (t_m 0.7-0.8).

Examples of logging impact on FGRs from tropical forests

Evidence of logging effects on FGRs from tropical forest tree species is relatively scarce and, when found, presents mixed results. Loss of genetic diversity has occurred in some logged populations of tropical timber species in Southeast Asia, but not in others (Wickneswari *et al.* 1997a and b). Both *Shorea leprosula* and *Dryobalanops aromatica* in Borneo showed high rates of outcrossing in both logged and unlogged forests (Kitamura *et al.* 1994). In a study of the self-incompatible species *Shorea siamensis* (Ghazoul *et al.* 1998) in Thailand, there were no differences noted in seed set following a moderate level of logging (35% removed) when compared to unlogged forest (Table 1). Although pollination levels in *S. siamensis* were similar before and after high-intensity logging, a lower level of intertree movements by pollinators in the more open, postlogging environment increased the frequency of self pollination, resulting in much lower fruit set.

Table 1. Fruiting of *Shorea siamensis* in Thailand under different disturbance levels.
Source: Ghazoul *et al.* 1998.

Fruiting parameter	Heavy disturbance	Moderate disturbance	Undisturbed
Flowering trees /ha	9	62	96
Fruit set 1996 (%)	0.7	2.2	2.5
Fruit set 1997 (%)	1.5	5.5	5.5
Pollinated flowers (%)	62	59	79

Research on how mating varies with tree or flowering-tree densities in natural forests and on the genetic effects of reduced tree densities under fragmentation will help infer the genetic impact of logging practices that reduce the density of a species of interest.

Several tropical tree species have different rates of outcrossing, both throughout the year and between individuals, that are reported as correlating with changes in the quantity of flowering and the spatial patterns of flowering individuals (Murawski and Hamrick 1991). Species occurring at low densities appear to combine some biparental mating with long-distance gene flow, whereas species occurring at higher densities showed more random mating, generally over shorter distances.

Mating patterns in three tropical tree species (*Calophyllum longifolium*, *Spondias mombin* and *Turpinia occidentalis*) that occur naturally at low densities in Panama were strongly affected by the spatial distribution of reproductive trees, although all three species still showed high levels of outcrossing. Where the trees were clumped, the majority of matings were with near neighbours, whereas with more evenly spaced trees a larger proportion of matings occurred between trees separated by distances of more than several hundred metres, and from well beyond the nearest reproductive neighbours (Stacy *et al.* 1996). In contrast, Murawski *et al.* (1990) and Murawski and Hamrick (1992b) demonstrated that outcrossing rates in the self-compatible species *Cavanillesia platanifolia* decreased in line with lower flowering levels from year to year (t_m 0.57 with 74% trees flowering, 0.35 with 49%, 0.21 with 32%). In years of greater flowering, with more floral rewards available, there was a greater tendency for pollinators to move among trees, resulting in more cross-pollination. However, when few trees were flowering there was a greater tendency for self-pollination. Studies of *Ceiba pentandra* in South America demonstrated variable degrees of self-fertility, with some trees showing high levels of self-pollination in isolation, while others failed to set seed (Murawski and Hamrick 1992a; Gribel *et al.* 1999). Therefore, the effects of logging on this species range from increased inbreeding to reduced seed production.

Variability of FGR responses to logging across different species

Genetic evidence from fragmentation and logging studies worldwide suggests that increases in inbreeding are especially dependent on the presence and strength of a species' incompatibility mechanism. Genetic variation can be conserved by reducing the frequency of inbreeding in naturally outcrossing tree species, while maintaining breeding system flexibility will be a conservation priority for species such as *Ceiba pentandra* that naturally combine outcrossing and inbreeding. Self-incompatible species like *Shorea siamensis* that show highly asynchronous population flowering may be more susceptible to reductions in population size following logging, both in terms of compatible pollination and reduced genetic diversity (Ghazoul *et al.* 1998), thereby reducing their regenerative capacity and their adaptive potential.

Tree species with specialist pollinators are more likely to face threats from reductions in both pollinator and tree populations as successful pollination becomes a limiting factor.

Table 2. Types of species potentially under threat of genetic erosion from low-impact selective logging. After Jennings *et al.* 2001.

Nature of species	Information source	Management solution
Commercial-sized stems form a large proportion of the species' population	Forest inventory data	Silvicultural treatments and further population inventories to ensure adequate regeneration before logging begins
Slow-growing species that reproduce only when large	Permanent sample plots and phenological studies	Discover minimum size at which trees become reproductive. Ensure there are more than 50 trees greater than this size in any compartment before next cut
Species with specific pollinators or seed dispersers	Published scientific literature. Studies of floral and fruit morphology, and animal vectors	Modify management to avoid or reduce habitat changes that will seriously affect pollinator or disperser populations
Rare or endemic species	Published floras and forest inventories	Do not log unless there are >500 reproductively mature individuals present in a contiguous forest area
Outcrossing species that are self-compatible	Reproductive biology research	Ensure that there are more than 50 reproductively mature trees in a compartment before the next cut

In contrast, species with unspecialized animal pollination regimes are only likely to suffer problems when there is a general loss of pollinator fauna in the immediate area where they grow. Table 2 summarizes possible characteristics of species that may be genetically susceptible to selective logging (Jennings *et al.* 2001).

Nevertheless, it is useful to consider that increased levels of inbreeding may be genetically unimportant from a long-term adaptation perspective, as self-pollinated individuals are selected against at various stages of regeneration. For selective logging, Ledig's (1992) assertion that "local genetic structure may be altered by selection and by changes in demography and in the mating system, but, in all probability, gene diversity and geographic structure will be little affected", suggests that if postlogging regeneration of rare or commercial species is adequate, their population genetic structure and diversity will not be severely or permanently affected. Genetic variation and the systems that maintain it are adaptable by nature, providing a degree of buffering and adaptation to change, but this is true only if changes are neither too great nor too rapid. Increased inbreeding is likely to be both genetically and ecologically important if it results in a reduction in seed production and, hence, in regeneration; or if it is maintained over several generations.

2.3 Tree planting

Where trees are planted, the levels of genetic diversity maintained will depend on the species involved and the germplasm collection practices. Tree planting can have a positive or negative effect on the genetic diversity of a plantation. Any existing level of inbreeding or reduction in genetic diversity in the sampled populations may have critical repercussions for plantations, on *ex situ* conservation or tree breeding programmes. Apart from the exceptions for self-compatible species already outlined, genetic diversity in trees is likely to be maintained when normal seed collecting protocols from natural populations are followed (Schmidt 2001). However, in the real world, departures from such best practice are not

uncommon. With species that produce large quantities of seed per tree, there is a tendency to make collections from a limited part of the crown and from a small number of trees, leading to reduced and biased sampling of the gene pool (Boshier *et al.* 1995). This can also occur unknowingly, especially when a pollen pool is dominated by a few trees (Aldrich and Hamrick 1998), and can lead to fewer genetic benefits than might be expected, even when conservation-motivated protocols are used.

Planting and using exotic trees or tree species at the expense of trees that have been either naturally regenerated or planted can also affect genetic conservation of local populations. For example, replacing native species or populations with introduced species or populations can reduce native population sizes, cause genetic contamination of native populations, and even eliminate native populations (Eldridge 1998; Hughes 1998). Hybridization of introduced tree species with native species is particularly prevalent in certain genera such as *Leucaena* (Hughes 1998) and *Prosopis* (Carney *et al.* 2000), and can have serious implications for the conservation of native gene pools, especially when threatened or endangered species are involved.

Crossings among populations of the same tree species from different locations can lead to the break-up of co-adapted allelic complexes or the dilution of adapted alleles, resulting in reduced growth or fertility and in outbreeding depression (Ledig 1992). Evidence in trees of outbreeding depression is conflicting and inconclusive within the few studies done thus far. For example, *Bombacopsis quinata* showed reduced seed set when populations from Honduras and Colombia were crossed (Billingham 1999), while Stacy (1998) similarly found that species of *Syzygium* and *Shorea* in southwestern Sri Lanka showed substantial reductions in fruit set following crosses involving pollen donors located some 12 km apart. However this author believed that the small-scale outbreeding depression found was more likely due to spatial heterogeneity in the selective environment than to isolation by geographic distance, as the geographic heterogeneity of the study area appeared to be more finely partitioned than that of many other tropical forest landscapes. In other cases, such as those described by Hardner *et al.* (1998) who undertook experiments where they both self-pollinated and crossed individuals of *Eucalyptus globulus* ssp. *globulus* (from distances of 21 m to 100 km) in Tasmania, no outbreeding depression was found. Neither was there evidence found in *Swietenia humilis* when populations in Central America 500 km apart were crossed (Billingham 1999).

3. Tools for monitoring the status and conservation of genetic diversity

Effective actions for the conservation of FGRs must be wide ranging and complementary at gene, population, species, community and ecosystem levels, and they must incorporate both *in situ* and *ex situ* strategies inside and outside reserves. Resource limitations dictate the need for integrated and simplified approaches. Consequently, in recognizing the importance of genetic considerations in achieving sustainable management, forest managers are now developing strategies that integrate the conservation of genetic diversity within production systems (Riggs 1990; Kuusipalo and Kangas 1994). It is now recognized that knowledge of the relationships between a species' genetic diversity, the heterogeneity of its habitat and the scale of its adaptation is important for understanding the evolutionary and ecological processes of the species and its associated biotic communities. This knowledge is also fundamentally important for the conservation of the species (e.g., priority setting for genetic reserves, sampling for *ex situ* collections, sourcing of material for use in ecological restoration), its successful reproduction, and its future adaptation to changing environmental conditions. Thus from both conservation and use viewpoints, it is necessary to know the extent of genetic variation within a species (allelic richness), how genetic variation is distributed (allelic evenness) and which geographic area retains the necessary and sufficient genetic variation to constitute a viable population in the long term.

Over the last 30 years, genetic markers such as allozymes and, more recently, a variety of DNA markers have permitted direct study of the distribution of genetic diversity within and between populations. For instance, a difference in the inheritance pattern of molecular

markers permits comparisons among relative levels of gene flow resulting from differential pollen and seed dispersal. Current data indicate greater interpopulation differentiation for maternally inherited genes than for nuclear genes in tree species (El Mousadik and Petit 1996; Hamrick and Nason 1996). Tests have also been developed that discriminate recently bottlenecked populations from stable populations (BOTTLENECK programme, see Cornuet and Luikart 1996; Luikart and Cornuet 1998). A small population size may be unimportant in a population that has always been characterized by low numbers, but it may be of fundamental importance to one that has only recently become small. While long-term demographic studies may provide a good understanding of long-term trends, rapid, informed solutions are required where populations face immediate threat. As demographic properties of populations influence their genetic make-up, so too can information from genetic markers go beyond the narrow study of marker diversity to provide estimates of demographic parameters of interest to conservation biologists.

Genealogical analyses of DNA sequence data can also be used to quantify important demographic parameters (e.g., effective population size, mating system or migration rate). Owing to the historical nature of such sequence information, a genealogical analysis reflects the long-term demographic properties of the population and is therefore useful in inferring its future demography (Milligan and Strand 1996). Comparison of such long-term estimates and current demographic observations can be used to determine if and how current trends deviate from historical ones, thereby providing information essential to the evaluation and management of tree species.

Molecular markers are now also being used to study phytogeographical patterns of variation in forest trees on broader scales. An example is the EU-funded CYTOFOR project (<http://www.pierroton.inra.fr/Cytofor>) that is investigating a number of genetic and ecological features of 22 economically important deciduous European trees through phylogeographic studies, the history of postglacial migration routes, assessment of hybridization between species, and the establishment of links between ecology and a population's genetic traits.

This research has the potential to generate information on the locations of greatest genetic diversity for these species, to guide EU policy on exchange of forest reproductive materials, and to help in devising conservation strategies for the species based on best ecological and genetic science. In addition, CYTOFOR results could improve understanding of the true 'origin' of forests that were established with non-native materials so long ago that this kind of information could not be found in conventional records. In most cases, *in situ* species conservation cannot ultimately succeed without the conservation of associated ecosystems; investigating several interacting species simultaneously promotes conservation of both ecosystems and individual species. Moreover, information on the location of older 'evolutionary units' for each of the 22 tree species, combined with results obtained from provenance tests, has immediate practical implications, since, if needed, recommendations can be made to restrict genetic movement between these evolutionary units.

4. Development of genetic criteria and indicators in sustainable forest management

The 1990s saw the development of a number of international sets of standards, criteria and indicators with which the sustainability of forest management could be assessed and certified. For use by auditors and managers in the field, these standards are often presented as checklists that give the principles, criteria, indicators and verifiers in a logical and progressive sequence, and from which it should be relatively easy to derive an objective decision on the quality of the forest management under assessment (e.g., Ghana's Forest Management Certification Standards and Checklist, Anon. 2000).

Almost all current international sets of standards pertaining to sustainable forest management include criteria and indicators that can be used to manage and assess the conservation of genetic diversity in trees (Box 1). The function of criteria and indicators is to provide a practical means by which changes in forest condition, as a consequence of management actions, can be monitored. Thus, criteria and indicators should be defined

Box 1. Some certification initiatives for sustainable forest management that mention criteria and indicators for measuring and/or assessing within-species genetic diversity

- 1. The Amazon Cooperation Treaty** (the ‘Tarapoto Process’; Grayson and Maynard 1997) is an international agreement subscribed to by eight South American countries in 1995 that seeks to ensure the long-term sustainability of forests in the Amazon. Criterion 4, Conservation of Forest Cover and of Biological Diversity, provides ‘measures for the conservation of genetic resources’ as an indicator of sustainability.
- 2. The International Tropical Timber Organization criteria and indicators for sustainable management of natural tropical forests** (ITTO 1998). Criterion 5, Biological Diversity, specifies a number of indicators, including 5.6, ‘Existence and implementation of a strategy for *in situ* and/or *ex situ* conservation of the genetic variation within commercial, endangered, rare and threatened species of forest flora and fauna.’
- 3. The Centre for International Forest Research, Criteria and Indicators Toolbox** (CIFOR C&I Team 1999). Criterion 2.3 requires the conservation of processes that maintain genetic variation. Indicators specify that there be no directional change in genotypic frequencies, no changes in gene flow/migration and no changes in mating systems.
- 4. Forest Stewardship Council Principles and Criteria¹**. Principle 6.3 requires that ecological functions and values be maintained intact, enhanced, or restored, and include criteria or principles for: (i) forest regeneration and succession; (ii) genetic, species, and ecosystem diversity; and (iii) natural cycles that affect the productivity of the forest ecosystem. Principle 9.3 requires that the management plan include and implement ‘specific measures that ensure the maintenance and/or enhancement of the applicable conservation attributes consistent with the precautionary approach’.

¹Forest Stewardship Council (FSC) 2004. Principles and Criteria. Document FSC-STD-01-001, http://www.fsc.org/fsc/whats_new/documents/Docs_cent/2,16

and presented in ways that are clear, practical and easy to use. They should also be based as far as possible on good science (ITTO 1998).

Unfortunately, the vital requirements of clarity, simplicity and practicality do not appear to have been seriously considered in the formulation of many of the genetic criteria and indicators developed to date for the management and monitoring of forest resources. For instance, all 23 verifiers (demographic and genetic) of the maintenance of genetic diversity proposed by Namkoong *et al.* (2002) require lengthy field trials or detailed monitoring of phenology, population structure, pollination and seed dispersal (Box 2, overleaf). Some even require sophisticated laboratory analyses that are impractical in day-to-day forest management. Several Namkoong verifiers involve the measurement of genetic parameters that are still the subject of scientific debate and that will produce values that vary according to the measurement method used. Decisions on baselines from which change and critical threshold values can be gauged are also problematical. In our view, such an overly theoretical and technical approach is not practical at the forest management unit level nor even at the country level when planners, owing to lack of information, face enormous real-world problems in designating nonloggable species and in determining other sustainable forest management strategies.

It is also our view that the criteria and indicators that incorporate genetic parameters to measure levels of genetic diversity conservation and the consequent sustainability of forestry operations, such as those proposed by CIFOR, are impractical and will in fact deter forest managers from making meaningful attempts at maintaining genetic diversity in production

Box 2. Proposed indicators and verifiers of the maintenance of genetic diversity in sustainable forest management

Indicator 1: Levels of variation

Demographic verifiers

- | | |
|-------|--|
| 1.D.1 | Census number of sexually mature individuals |
| 1.D.2 | Census number of reproducing individuals |
| 1.D.3 | Coefficient of phenotypic variation |

Genetic verifiers

- | | |
|-------|-------------------|
| 1.G.1 | Number of alleles |
| 1.G.2 | Gene diversity |
| 1.G.3 | Genetic variation |

Indicator 2: Directional change in allele or genotype frequencies

Demographic verifiers

- | | |
|-------|-----------------------|
| 2.D.1 | Phenotypic shifts |
| 2.D.2 | Age/size class shifts |
| 2.D.3 | Environmental shifts |

Genetic verifiers

- | | |
|-------|----------------------------|
| 2.G.1 | Genotypic frequency shifts |
| 2.G.2 | Marker frequency shifts |
| 2.G.3 | Genetic mean shifts |

Indicator 3: Migration among populations

Demographic verifiers

- | | |
|-------|--------------------|
| 3.D.1 | Physical isolation |
| 3.D.2 | Mating isolation |
| 3.D.3 | Seed dispersal |
| 3.D.4 | Pollen dispersal |

Genetic verifiers

- | | |
|-------|-----------|
| 3.G.1 | Gene flow |
|-------|-----------|

Indicator 4: Reproductive system

Demographic verifiers

- | | |
|-------|----------------------|
| 4.D.1 | Parental pool size |
| 4.D.2 | Seed germination |
| 4.D.3 | Pollinator abundance |
| 4.D.4 | Sexuality |

Genetic verifiers

- | | |
|-------|-------------------|
| 4.G.1 | Outcrossing rate |
| 4.G.2 | Correlated mating |

Source: Namkoong *et al.* 2002.

forests (Jennings *et al.* 2001). In our opinion, the impractical nature of such parameters explains why the Ghana Checklist omits mention of within-species genetic diversity in its principles, and only implicitly addresses conservation of a species' genetic diversity within coarse grain protection. In fact, the Ghana Checklist does not contain genetic verifiers, whether in the form of genetic assessments or of surrogate demographic parameters. If the maintenance of genetic diversity is not to be completely excluded in undertakings like the Ghana Checklist, a more pragmatic approach needs to be developed that is based on existing knowledge of how logging affects genetic processes. The work coordinated by the International Plant Genetic Resources Institute (IPGRI), in collaboration with the Amazon Institute of People and the Environment (IMAZON) and the Institute of Forestry and Agricultural Management and Certification (IMAFLOA) in the Brazilian Amazon, is one attempt to draw general conclusions from already certified forest plots on the feasibility of using genetic information in certification schemes (IMAFLOA/IMAZON/IPGRI 2004).

5. Tools for conservation actions

In order to establish the significance of observed levels of genetic diversity and inbreeding, baseline information is required from which deviations from the norm can be observed. However, studies to establish baseline data are often compromised by the inability of researchers to decide on a realistic baseline population from which changes can be observed and significance analyzed. Comparative studies of human disturbance are similarly hampered by the range of experimental designs and markers employed. Furthermore, problems in interpretation of results are compounded by both the diversity of management systems used around the world and the variety of reproductive and ecological regimes characteristic of the species under study. Most studies have looked only at the effects on genetic diversity immediately after logging or over one rotation, while several generations may normally be necessary before genetic erosion becomes evident. The cost and difficulty of carrying out such long-term studies requires the use of other approaches.

ECO-GENE is a model that has been developed to study the impact of silvicultural actions on temperate forests (Degen *et al.* 1996). It combines research on population genetic and dynamic processes with the use of forest growth models to simulate the impact of management types and intensities on within-species genetic diversity. Adaptation and validation of such temperate models in more complex tropical forest conditions require extensive reproductive ecology and genetic data on the selected species being modelled (Degen *et al.* 2002; see also Chapter 4 in this book). Knowledge of a species' reproductive ecology, including phenology, incompatibility mechanism, pollination mechanism, dispersal, seed dispersal, seedling regeneration and stand dynamics enables researchers to predict how tree removal by logging will affect the subsequent capacity of that species to regenerate. The regeneration phase is critical not only to the maintenance of a forest, but also to assessing the genetic implications of increased levels of inbreeding for sustainable management. Although information exists on the reproductive ecology, genetics and mating patterns for a number of tropical tree species, there are relatively few studies on the effects of different management strategies on these processes. Use of existing genetic data from unmanaged or undisturbed forests is unlikely to produce valid comparative models and thereby help identify critical scenarios that will allow accurate predictions. The validation of models and their subsequent ability to predict accurately the impacts of management on genetic diversity will therefore depend on the generation of new data that estimate pollen flow within populations under logging conditions. Recommendations can then be made on a species-by-species basis that will help prevent managers from crossing subjective genetic thresholds by determining permissible levels of logging and patterns of tree selection that minimize inbreeding in species identified as problematical.

It is also important to recognize that tree species have developed a range of reproductive strategies, and that the conservation of trees can be adversely affected by management approaches that dramatically alter these processes. Thus, reducing the level or impact of inbreeding while maintaining diversity in naturally outcrossing tree species will be important to their conservation. Maintaining breeding system flexibility will also be a priority for species that naturally combine outcrossing and inbreeding. However, as many commercial timber species fall into the ecological groupings of 'pioneer' or 'long-lived shade-intolerant' trees, the choice of species to model should not be limited to commercial species, but should also take into account other ecological guilds with different mating systems and spatial distribution. By their nature, many pioneer or long-lived shade intolerant trees are to some degree genetically resilient to disturbance. A species will most likely be genetically suitable to management if it:

- has low population densities; is slow growing
- is shade-intolerant
- is outcrossing but is self-compatible
- produces occasional seed
- is late-maturing

- regenerates poorly under human disturbance
- possesses specialized pollinators or seed dispersers
- is characterized by clumped distributions (Jennings *et al.* 2001).

Inevitably, generalizations like this will be qualified by the range of factors that have been shown to influence genetic variation in trees, but using a validated model to identify factors that leave species genetically favourable to management will allow species to be grouped into management groupings and for guidelines to be extended to a wider range of species. The ability to extrapolate from modelling results and to make more general management recommendations for groups of species will depend on accurate information that will enable classification of species into such management groups. An attempt has been made to model species conservation decision-making within a framework of goal conflicts and uncertainty (Drechsler 2004), but this was done without genetic considerations.

Limited resources and the lack of ecological and genetic information, along with an immediate need to generate the measures required for certification of sustainable logging in many parts of the world, has led to pragmatic 'best guess' approaches based on available knowledge. Adopting simple silvicultural rules before felling is permitted helps to ensure that adequate natural regeneration will occur after logging, which can also keep the effects of logging on genetic diversity to a minimum. Identifying the types of species and the circumstances (whether ecological, managerial or other) under which simple silvicultural rules cannot be developed or will not work should be a priority for managers with limited research resources. This will force them to undertake realistic assessments of the species that seem to be at genetic risk from disturbance and to develop management strategies to alleviate possible problems (Table 2). Thus, reasonable questions for a certifier to ask a forest manager or a country's forest authority are: 'Which species are genetically threatened by logging?' 'What is the basis for this determination?' and 'What remedial actions have you put in place to conserve these species?'

Managers must similarly make decisions about the conservation benefits of land-use alternatives. Several of the studies discussed here are optimistic about the conservation value of remnant trees and forests. In cases where large tracts of forest no longer exist, conservation initiatives have emphasized connectivity of landscape mosaics. Such examples illustrate a broader, integrated vision of conservation that embraces a range of land-use mosaics rather than just intact forests. Design, management and monitoring of forests and forest reserves will thus involve assessment of the managed land-use types in terms of how well – either individually or in combination – they meet the biological criterion of connectivity, as well as other genetic and ecological requirements, and how these land-use types may need to be modified to maintain or improve connectivity and diversity (Laurance *et al.* 1997). Nevertheless, assessments of this kind will still need to contain site-specific elements, given the varying connectivity, species management and sustainable land-use objectives of different people and organizations, and the different degrees of resilience of forests and their components to disturbance. In an area of high forest cover, agroecosystems may be valued principally for gene flow, whereas in much more highly deforested landscapes a fuller complement of biological benefits may be sought from agroecosystems, with their specific location within the corridor that connects different areas also of importance, depending on the species they contain. For example, in the highly deforested dry forest zone of western Honduras, the traditional Quezungual fallow system, which includes management of naturally regenerated shrubs, fruit trees and timber trees along with crops (Kass *et al.* 1993), is likely to provide a variety of genetic conservation benefits for a range of native tree species. Other complex agricultural systems, such as traditional shaded coffee, may also rate highly for genetic conservation benefits. In contrast, simpler systems with only a few pastureland trees may offer fewer genetic conservation benefits, and are unlikely to prove effective mediators of pollen flow for species without self-incompatibility mechanisms.

Therefore, where appropriate, management of forests and tree species must also take into account the farming systems practiced in the area along with the density of the trees and their origin and whether regeneration is natural or planted. For example,

the maintenance of native timber trees as an overstorey in large areas that have been converted to coffee plantations is likely to have beneficial genetic effects for gene flow, population numbers and the conservation of those tree species. In contrast, where such a management system is practised over only a small area, seed production may show reduced genetic diversity through related or biparental mating (Boshier *et al.* 2004).

Gap analysis, a scientific means for assessing to what extent native animal and plant species are being protected, provides an appropriate tool to integrate such information and to examine the effectiveness of existing land-use mosaics in conserving the genetic resources of target species. Recent studies of eight native conifer tree species in the US Pacific Northwest stratified species distribution into genetic conservation units, and employed gap analysis to identify those units for which the genetic resources were not well-conserved (i.e., fewer than 5000 reproductive individuals) in existing protected areas (Lipow *et al.* 2004).

6. Conclusion

Documenting and assessing human impact on genetic diversity in forest trees are challenging. The diversity of tree species' biology and genecology means that species will respond in varying ways to different types of human disturbance and, therefore, assessing causal relationships and direct impacts cannot be easily generalized. The remaining chapters in this book provide new examples of human effects on species with different biological characteristics.

Nevertheless, some patterns are evident from the studies reviewed in this chapter and as such they have implications for actions to conserve FGRs. This is well illustrated by the issue of remnant forest fragments and trees on farms. The evidence summarized in this chapter suggests that trees that are present in a range of complex landscapes and agroecosystems may in fact play important roles in the long-term genetic viability of many native tree species. They will do this by facilitating gene flow among forest fragments, by conserving genotypes not found in reserves or fragments, by helping to maintain minimum viable populations, and by acting as intermediaries and alternative hosts for pollinators and seed dispersers (Harvey and Haber 1999), thereby functioning as biological corridors. Thus, it is important to recognize the complementary role that maintaining trees on farms can play in conserving FGRs.

Despite this assertion, and although trees maintained in agricultural systems undoubtedly contribute to reproduction in remnant forests, the benefits and effects are complex and vary from species to species. Thus, unevenness and over-representation of such trees in pollen pools may lead to nonrandom mating and reductions in genetic diversity in subsequent generations. The use of trees growing in agricultural systems as seed sources for ecological restoration is under debate, especially in situations where 'local' seed sources, presumed to be adapted to local conditions, may have problems of low genetic diversity and inbreeding.

Therefore, we should neither over- nor underestimate the extent to which trees growing in agroecosystems can benefit the genetic conservation of forest tree species. This is especially true for the many tree species found within agroecosystems that also occur in adequate numbers in existing forest fragments. On the other hand, some of the species threatened by low population numbers, either as a result of deforestation, fragmentation, logging practices or natural rarity, will not be found widely in agroecosystems, since in these domesticated environments conditions are unsuited to their regeneration. Thus, the greatest potential conservation role for trees in agroecosystems will be in highly deforested areas where forest fragments or reserves are small or nonexistent and where safeguarding these trees represents an important part of the gene pool of a particular population or species. Under these circumstances, the conservation of tree species in domesticated landscapes, often under the auspices of traditional management practices, can free resources for more critically threatened tree species that require more conventional, resource-intensive conservation approaches, usually involving *in situ* methods. Underestimating the capacity of many species to persist in large numbers in

such agroecosystems could lead to the misdirection of limited conservation resources towards species not actually under threat (Boshier *et al.* 2004).

The benefits of different land-use practices for the conservation of FGRs need to be recognized, promoted and, of course, evaluated on a continuing basis. Development and implementation of conservation strategies in areas of human disturbance demand an interdisciplinary approach that balances conservation requirements with basic human needs. We must raise awareness among development organizations that natural regeneration can have conservation benefits and also provide the socioeconomic resources favoured by local inhabitants. We should also seek close consultation with stakeholders to determine not only the potential for integrating conservation and economic development, but also which species could be conserved in these systems from both biological and use perspectives. This will require the involvement of development organizations in biodiversity conservation and effective two-way communication between the two groups. Collaboration like this will ensure both conservation and development benefits, and include the deployment of locally adapted, diverse tree germplasm to supply the multipurpose requirements of goods and services for people.

References

- Aldrich, P.R. and J.L. Hamrick. 1998. Reproductive dominance of pasture trees in a fragmented tropical forest mosaic. *Science* 281:103-105.
- Allen-Wardell, G., P. Bernhardt, R. Bitner, A. Burquez, S. Buchmann, J. Cane, P.A. Cox, V. Dalton, P. Feinsinger, M. Ingram, D. Inouye, C.E. Jones, K. Kennedy, P. Kevan, H. Koopowitz, R. Medellin, S. Medellin-Morales and G.P. Nabhan. 1998. The potential consequences of pollinator declines on the conservation of biodiversity and stability of food crop yields. *Conserv. Biol.* 12:8-17.
- Anon. 2000. Forest Management Certification Standards and Checklist (version 4) of the Ghana Forest Management Certification Systems Project (EU Contract B7-620/97-/4VIII/ FOR and NEDA Project GH 008701).
- Apsit, V. and J.L. Hamrick. (In review). Estimating pollen flow and effective population sizes of fragmented *Enterolobium cyclocarpum* populations in Costa Rica. Submitted to *Evolution*.
- Ballal, S.R., S.A. Foré and S.I. Guttman. 1994. Apparent gene flow and genetic structure of *Acer saccharum* subpopulations in forest fragments. *Can. J. Bot.* 72:1311-1315.
- Billingham, M.R. 1999. Genetic structure, localised adaptation and optimal outcrossing distance in two neotropical tree species. PhD Thesis, University of Oxford, UK.
- Boshier, D.H., M.R. Chase and K.S. Bawa. 1995. Population genetics of *Cordia alliodora* (Boraginaceae), a neotropical tree. 2. Mating system. *Am. J. Bot.* 82:476-483.
- Boshier, D.H., J.E. Gordon and A.J. Barrance. 2004. Prospects for circa situm tree conservation in Mesoamerican dry forest agro-ecosystems. Pp. 210-226 in *Biodiversity Conservation in Costa Rica, Learning the Lessons in the Seasonal Dry Forest* (G.W. Frankie, A. Mata and S.B. Vinson, eds.). University of California Press, Berkeley, USA.
- Brown, A.H.D. and C.M. Hardner. 2000. Sampling the gene pools of forest trees for *ex situ* conservation. Pp. 185-196 in *Forest Conservation Genetics: Principles and Practice* (A. Young, D.H. Boshier and T.J. Boyle, eds.). Commonwealth Scientific and Industrial Research Organization (CSIRO) Publishing, Melbourne, Australia and CAB International, Wallingford, UK.
- Buchert, G.P., O.P. Rajora, J.V. Hood and B.P. Dancik. 1997. Effects of harvesting on genetic diversity in old-growth eastern white pine in Ontario, Canada. *Conserv. Biol.* 11:747-758.
- Carney, S.E., D.E. Wolf and L.H. Rieseberg. 2000. Hybridisation and forest conservation. Pp. 167-182 in *Forest Conservation Genetics: Principles and Practice* (A. Young, D.H. Boshier and T.J. Boyle, eds.). Commonwealth Scientific and Industrial Research Organization (CSIRO) Publishing, Melbourne, Australia and CAB International, Wallingford, UK.
- CIFOR C&I (Centre for International Forestry Research Criteria and Indicators) Team. 1999. CIFOR Criteria and Indicators Generic Template. Criteria and Indicators Toolbox Series No. 2. CIFOR, Bogor, Indonesia.

- Cornuet, J.M. and G. Luikart. 1996. Description and power analysis of two tests for detecting recent population bottlenecks from allele frequency data. *Genetics* 144:2001-2014.
- Degen, B., H.-R. Gregorius and F. Scholz. 1996. ECO-GENE, a model for simulation studies on the spatial and temporal dynamics of genetic structures of tree populations. *Silvae Genet.* 45:323-329.
- Degen, B., D.W. Roubik and M.D. Loveless. 2002. Impact of selective logging and forest fragmentation on the seed cohorts of an insect-pollinated tree: a simulation study. Pp. 108-119 *in* Modelling and Experimental Research on Genetic Processes in Tropical and Temperate Forests, (B. Degen, M.D. Loveless and A. Kremer, eds.). EMBRAPA (Empresa Brasileira de Pesquisa Agropecuária) Amazonia Oriental, Belém, Brazil.
- Dick, C.W. 2001. Genetic rescue of remnant tropical trees by an alien pollinator. *Proc. R. Soc. Lond. B. Biol. Sci.* 268:2391-2396.
- Dick, C.W., G. Etchelecu and F. Austerlitz. 2003. Pollen dispersal of tropical trees (*Dinizia excelsa*: Fabaceae) by native insects and African honeybees in pristine and fragmented Amazonian rainforest. *Mol. Ecol.* 12:753-764.
- Drechsler, M. 2004. Model-based conservation decision aiding in the presence of goal conflicts and uncertainty. *Biodiv. Conserv.* 13:141-164.
- El Mousadik A. and R.J. Petit. 1996. High level of genetic differentiation for allelic richness among populations of the argan tree *Argania spinosa* (L.) Skeels endemic to Morocco. *Theor. Appl. Genet.* 92:832-839
- Eldridge, K.G. 1998. Californian Radiata pine seed in store: what to do with it? Commonwealth Scientific and Industrial Research Organization (CSIRO) Forestry and Forest Products, client report no 389. Canberra, Australia.
- Eldridge, K., J. Davidson, C. Harwood and G. van Wyk (eds.). 1993. *Eucalyptus Domestication and Breeding*. Clarendon Press, Oxford, UK.
- Foré, S.A., R.J. Hickey, J.L. Vankat, S.I. Guttman and R.L. Schaefer. 1992. Genetic structure after forest fragmentation: a landscape ecology perspective on *Acer saccharum*. *Can. J. Bot.* 70:1659-1658.
- Friedman, S.T. and W.T. Adams. 1985. Estimation of gene flow into two seed orchards of loblolly pine (*Pinus taeda* L.). *Theor. Appl. Genet.* 69:609-615.
- Geburek, T. 2000. Effects of environmental pollution on the genetics of trees. Pp. 135-157 *in* Forest Conservation Genetics: Principles and Practice (A. Young, D.H. Boshier and T.J. Boyle, eds.). Commonwealth Scientific and Industrial Research Organization (CSIRO) Publishing, Melbourne, Australia and CAB International, Wallingford, UK.
- Ghazoul, J., K.A. Liston and T.J.B. Boyle. 1998. Disturbance induced density-dependent seed set in *Shorea siamensis* (Dipterocarpaceae), a tropical forest tree. *J. Ecol.* 86:462-473.
- Glaubitz, J.C., J.C. Murrell and G.F. Moran. 2003a. Effects of native forest regeneration practices on genetic diversity in *Eucalyptus consideniana*. *Theor. Appl. Genet.* 107:422-431.
- Glaubitz, J.C., H. Wu and G.F. Moran. 2003b. Impacts of silviculture on genetic diversity in the native forest species *Eucalyptus sieberi*. *Conserv. Genet.* 4:275-287.
- Grayson, A.J. and W.B. Maynard (eds.). 1997. *The World's Forests – Rio+5: International Initiatives towards Sustainable Management*. Commonwealth Forestry Association, Oxford, UK.
- Gribel, R., P.E. Gibbs and A.L. Queiroz. 1999. Flowering phenology and pollination biology of *Ceiba pentandra* (Bombaceae) in central Amazonia. *J. Trop. Ecol.* 15:247-263.
- Griffin, A.R. 1990. Effects of inbreeding on growth of forest trees and implications for management of seed supplies for plantation programmes. Pp. 355-374 *in* Reproductive Ecology of Tropical Forest Plants, (K.S. Bawa and M. Hadley, eds.). Parthenon Publishing, London, UK.
- Hamrick, J.L. 1992. Distribution of genetic diversity in tropical tree populations: implications for the conservation of genetic resources. Pp. 74-82 *in* Breeding Tropical Trees, Proceedings of the International Union of Forest Research Organizations (IUFRO) S2.02-08 Conference 9-18 October 1992, Cali, Colombia.

- Hamrick, J.L. and J.D. Nason. 1996. Consequences of dispersal in plants. Pp. 203-236 in *Population Dynamics in Ecological Space and Time*, (O.E. Rhodes, R.K. Chesser and M.H. Smith, eds.). University of Chicago Press, Chicago, USA.
- Hardner, C.M., B.M. Potts and P.L. Gore. 1998. The relationship between cross success and spatial proximity of *Eucalyptus globulus* ssp. *globulus* parents. *Evolution* 52:614-618.
- Harvey, C.A. and W.A. Haber. 1999. Remnant trees and the conservation of biodiversity in Costa Rican pastures. *Agrof. Systems* 44:37-68.
- Hawthorne, W.D., V. Agyeman and M. Abu Juam. 1999. Improving the logging system in Ghana: research results from Department of International Development Forestry Research Programme Project R6716. Pp. 89 in *Workshop Proceedings*, Kumasi, Ghana, 29-30 March 1999. FRR, Bristol, UK.
- Henle, K., D.B. Lindenmayer, C.R. Margules, D.A. Saunders and C. Wissel. 2004. Species survival in fragmented landscapes: where are we now? *Biodivers. Conserv.* 13:1-8.
- Heywood, V.H. and S.N. Stuart. 1992. Species extinctions in tropical forests. Pp. 91-117 in *Tropical Deforestation and Species Extinction*, (T.C. Whitmore and J.A. Sayer, eds.). Chapman and Hall, London, UK.
- Hughes, C.E. 1998. *Leucaena: a Genetic Resources Handbook*. Tropical Forestry Paper No. 37, Oxford Forestry Institute, Oxford, UK.
- IMAFLOA/IMAZON/IPGRI (Institute of Forestry and Agricultural Management and Certification/Amazon Institute of People and the Environment/International Plant Genetic Resources Institute). 2004. Final report of the project Criteria and Indicators for Monitoring and Evaluation of the Genetic Sustainability of Forest Management. IPGRI, Rome, Italy.
- ITTO (International Tropical Timber Organization). 1998. Criteria and indicators for sustainable management of natural tropical forests. ITTO Policy Development Series No. 7.
- Janzen, D.H. 1986. Blurry catastrophes. *Oikos* 47:1-2.
- Jennings, S.B., N.D. Brown, D.H. Boshier, T.C. Whitmore and J.C.A. Lopes. 2001. Ecology provides a pragmatic solution to the maintenance of genetic diversity in sustainably managed tropical rain forests. *For. Ecol. Manage.* 154:1-10.
- Kass, D.C.L., C. Foletti, L.T. Szott, R. Landaverde and R. Nolasco. 1993. Traditional fallow systems of the Americas. *Agrof. Systems* 23:207-218.
- Kitamura, K., A.R. Mohamad Yusof, O. Ochiai and H. Yoshimaru. 1994. Estimation of outcrossing rate on *Dryobalanops aromatica* Gaertn. F. in primary and secondary forests in Brunei, Borneo, southeast Asia. *Plant Species Biol.* 9:37-41.
- Kuusipalo, J. and J. Kangas. 1994. Managing biodiversity in a forestry environment. *Conserv. Biol.* 8:450-460.
- Laurance, W.F., R.O. Bierregaard Jr, C. Gascon, R.K. Didham, A.P. Smith, A.J. Lynam, V.M. Viana, T.E. Lovejoy, K. Sieving, J.W. Sites Jr, M. Andersen, M. Tocher, E. Kramer, C. Restrepo and C. Moritz. 1997. Tropical forest fragmentation: synthesis of a diverse and dynamic discipline. Pp. 502-514 in *Tropical Forest Remnants: Ecology, Management, and Conservation of Fragmented Communities*, (W.F. Laurance and R.O. Bierregaard Jr, eds.). University of Chicago Press, Chicago, USA.
- Ledig, F.T. 1988. The conservation of genetic diversity in forest trees. *Bioscience* 38:471-479.
- Ledig, F.T. 1992. Human impacts on genetic diversity in forest ecosystems. *Oikos* 63:87-108.
- Lipow, S.R., K. Vance-Borland, J.B. St Clair, J. Henderson and C. McCain. 2004. Gap analysis of conserved genetic resources for forest trees. *Conserv. Biol.* 18:412-423.
- Luikart, G. and J.M. Cornuet. 1998. Empirical evaluation of a test for identifying recently bottlenecked populations from allele frequency data. *Conserv. Biol.* 12:228-237.
- Mátyás, C. 1996. Climatic adaptation of trees: rediscovering provenance tests. *Euphytica* 92:45-54.
- Melbourne, B.A., K.F. Davies, C.R. Margules, D.B. Lindenmayer, D.A. Saunders, C. Wissel and K. Henle. 2004. Species survival in fragmented landscapes: where to from here? *Biodivers. Conserv.* 13:275-284.

- Milligan, B.G. and A.E. Strand. 1996. Genetics and conservation biology: assessing historical trends in the demography of populations. Pp. 125-137 in *Southwestern Rare and Endangered Plants: Proceedings of the Second Conference*, (J. Maschinski, H.D. Hammond and L. Holter, eds.). General Technical Report RM-GTR-283. United States Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station, Fort Collins, USA.
- Murawski, D.A. and J.L. Hamrick. 1991. The effect of the density of flowering individuals on the mating systems of nine tropical tree species. *Heredity* 67:167-174.
- Murawski, D.A. and J.L. Hamrick. 1992a. The mating system of *Cavanillesia platanifolia* under extremes of flowering-tree density: a test of predictions. *Biotropica* 24:99-101.
- Murawski, D.A. and J.L. Hamrick. 1992b. Mating system and phenology of *Ceiba pentandra* (Bombacaceae) in central Panama. *J. Hered.* 83:401-404.
- Murawski, D.A., J.L. Hamrick, S.P. Hubbell and R.B. Foster. 1990. Mating systems of two Bombacaceous trees of a neotropical moist forest. *Oecologia* 82:501-506.
- Namkoong G. 1993. A gene conservation plan for loblolly pine. *Can. J. For. Res.* 27:433-437.
- Namkoong, G., T. Boyle, Y.A. El-Kassaby, C. Palmberg-Lerche, G. Eriksson, H.-R. Gregorius, H. Joly, A. Kremer, O. Savolainen, R. Wickneswari, A. Young, M. Zeh-Nlo and R. Prabhu. 2002. Criteria and indicators for sustainable forest management: assessment and monitoring of genetic variation. Forest Genetic Resources Working Paper FGR/37E, Forest Resources Development Service, Forest Resources Division, Forestry Department, Food and Agriculture Organization of the United Nations (FAO). Rome, Italy.
- Nason, J.D. and J.L. Hamrick. 1997. Reproductive and genetic consequences of forest fragmentation: two case studies of neotropical canopy trees. *J. Hered.* 88:264-276.
- Neale, D.B. 1985. Genetic implications of shelterwood regeneration of Douglas-fir in southwest Oregon. *For. Sci.* 31:995-1005.
- Neel, M.C. and M.P. Cummings. 2003. Effectiveness of conservation targets in capturing genetic diversity. *Conserv. Biol.* 17:219-221.
- Prober, S.M. and A.H.D. Brown. 1994. Conservation of the grassy white box woodlands – population genetics and fragmentation of *Eucalyptus albens*. *Conserv. Biol.* 8:1003-1013.
- Riggs, L.A. 1990. Conserving genetic resources on-site in forest ecosystems. *For. Ecol. Manage.* 35:45-68.
- Robinson, D. 1996. Participatory resource use studies: a tool for community forest management. Unpublished MSc Dissertation, University of Oxford, UK.
- Rocha, O.J. and G. Aguilar. 2001. Reproductive biology of the dry forest tree *Enterolobium cyclocarpum* (Guanacaste) in Costa Rica: a comparison between trees left in pastures and trees in continuous forest. *Am. J. Bot.* 88:1607-1614.
- Rowe, R. and Q. Cronk. 1995. Applying molecular techniques to plant conservation: screening genes for survival. *Plant Talk* 1:18-19.
- Saunders, D.A., R.J. Hobbs and C.R. Margules. 1991. Biological consequences of ecosystem fragmentation: a review. *Biol. Conserv.* 5:18-32.
- Schmidt, L. 2001. Guide to Handling of Tropical and Subtropical Forest Seed. Danish International Development Agency (DANIDA) Forest Seed Centre, Humlebaek, Denmark.
- Schmidtling, R.C. 1994. Use of provenance tests to predict response to climatic change: loblolly pine and Norway spruce. *Tree Physiol.* 14:805-817.
- Schnabel, A. and J.L. Hamrick. 1995. Understanding the population genetic structure of *Gleditsia triacanthos* L.: the scale and pattern of pollen gene flow. *Evolution* 49:921-913.
- Sim, B.L. 1984. The genetic base of *Acacia mangium* Willd. in Sabah. Pp. 597-603 in *Provenance and Genetic Improvement Strategies in Tropical Forest Trees*, (R.D. Barnes and G.L. Gibson, eds.). Commonwealth Forestry Institute, Oxford, UK, and Forest Research Centre, Harare, Zimbabwe.

- Stacy, E.A. 1998. Cross-compatibility in tropical trees: associations with outcrossing distance, inbreeding, and seed dispersal. *Am. J. Bot.* 85(6):62.
- Stacy, E.A., J.L. Hamrick, J.D. Nason, S.P. Hubbell, R.B. Foster and R. Condit. 1996. Pollen dispersal in low-density populations of three neotropical tree species. *Am. Nat.* 148:275-298.
- Thomas, B.R., S.E. Macdonald, M. Hicks, D.L. Adams and R.B. Hodgetts. 1999. Effects of reforestation methods on genetic diversity of lodgepole pine: an assessment using microsatellite and randomly amplified polymorphic DNA markers. *Theor. Appl. Genet.* 98:793-801.
- White, G.M., D.H. Boshier and W. Powell. 2002. Increased pollen flow counteracts fragmentation in a tropical dry forest: an example from *Swietenia humilis* Zuccarini. *Proc. Nat. Acad. Sci. USA* 99:2038-2042.
- Wickneswari, R. and T.J.B. Boyle. 2000. Effects of logging and other forms of harvesting on genetic diversity in humid tropical forests. Pp. 115-122 in *Forest Conservation Genetics: Principles and Practice*, (A. Young, D. Boshier and T.J. Boyle, eds.) CAB International, Wallingford, UK and Commonwealth Scientific and Industrial Research Organization (CSIRO) Publishing, Collingwood, Australia.
- Wickneswari, R., C.T. Lee, M. Norwati and T.J.B. Boyle. 1997a. Immediate effects of logging on the genetic diversity of five tropical rainforest species in a ridge forest in peninsular Malaysia. Paper presented at CIFOR wrap-up workshop on Impact of Disturbance, Bangalore, India, August 1997. Centre for International Forestry Research (CIFOR), Bogor, Indonesia.
- Wickneswari, R., C.T. Lee, M. Norwati and T.J.B. Boyle. 1997b. Effects of logging on the genetic diversity of six tropical rainforest species in a regenerated mixed dipterocarp lowland forest in peninsular Malaysia. Paper presented at CIFOR wrap-up workshop on Impact of Disturbance, Bangalore, India, August 1997. Centre for International Forestry Research (CIFOR), Bogor, Indonesia.
- Williams, C.G. and O. Savolainen. 1996. Inbreeding depression in conifers: implication for breeding strategy. *For. Sci.* 42:102-117.
- Young, A. and T. Boyle. 2000. Forest fragmentation. Pp. 123-134 in *Forest Conservation Genetics: Principles and Practice*, (A. Young, D. Boshier and T. Boyle, eds.) CAB International, Wallingford, UK and Commonwealth Scientific and Industrial Research Organization (CSIRO) Publishing, Collingwood, Australia.
- Young, A., D.H. Boshier and T.J. Boyle (eds.). 2000. *Forest Conservation Genetics: Principles and Practice*. Commonwealth Scientific and Industrial Research Organization (CSIRO) Publishing, Melbourne, Australia and CAB International, Wallingford, UK.
- Young, A., T.J. Boyle and T. Brown. 1996. The population genetic consequences of habitat fragmentation for plants. *Trends Ecol. Evol.* 10:413-418.
- Young, A., H.G. Merriam and S.I. Warwick. 1993. The effects of forest fragmentation on genetic variation in *Acer saccharum* Marsh. (sugar maple) populations. *Heredity* 71:277-289.

Chapter 2

Challenges and opportunities for communal forest management in South America

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1. Introduction

This chapter analyzes the principal factors that influence the successes and failures of communally managed forest resources in a range of South American contexts. It reviews and summarizes the findings of the growing body of literature that deals with this issue and then examines the findings in the context of case studies from five South American countries.

Most scholars who study the institutional arrangements for natural resource governance agree that forests should be considered as common-pool resources (CPRs) since they are neither public nor private goods, but share elements of both. This characteristic means that forests are particularly vulnerable to degradation and overexploitation because it is difficult to exclude people from them, leading to overconsumption (that is, they are 'subtractable' resources, see Section 2 below). In order to avoid such negative outcomes, and in order to manage forests sustainably, we argue that communities who manage forests need institutions. These are defined in this study as sets of agreed-upon rules that are followed by most community members and which control access to and regulate competition over forest resources (Winter 1998; Ostrom *et al.* 2002).

Section 2 of this chapter is devoted to understanding forests as CPRs by describing and discussing the main problems that local forest-user groups face in managing their forests. This section also reviews the literature on the principal factors, attributes and indicators that relate to the effectiveness of communal forest management, summarized as attributes either of the resource or of the resource user. Our discussion relies principally on research by Ostrom (1998), Gibson *et al.* (2000), and Agrawal (2001) that summarizes the challenges to community management of forests. These challenges result from particular attributes of the resource, in this case the forest, and of the users.

Section 3 provides a historical overview of some specific features of communal forest management in South America. When compared to formal community forestry groups in developing regions of East Africa and South Asia, South American forest-user groups have access to relatively valuable forest resources. While formal governmental legislation to enable community forestry is fairly recent in South America, the challenge for rural communities to govern their forests is not new. As a consequence, we predict that local institutions in South America will successfully mitigate the effects of exogenous factors such as national policies and market forces.

Section 4 of this chapter presents three short case studies that highlight how some rural communities have modified local institutions to deal with or to take advantage of new governmental regulations and market opportunities in the forestry sector. Case 1 comes from Bolivia where laws required indigenous Yuracaré communities to document their historical use of land before they could secure forest ownership. The Yuracaré also had to develop a forest management plan approved by the forestry service before they could harvest trees – even from their own land. Working with an International Forestry

Resources and Institutions (IFRI) team that included social scientists and foresters, the community was able to document its history and develop forest plans to gain these rights. As a consequence, the traditional subsistence uses of the forest were expanded to include harvesting for the market, and the Yuracaré had to develop new local institutions to deal with the new market opportunities (Becker and Leon 2000).

Case 2 comes from Mexico. Here, a Zapotec indigenous group possessed local institutions for managing forests for subsistence but reorganized them to take advantage of new rights to market timber with the Mexican government. To date, the community has developed forest management plans and received a new forestry certification from the SmartWood programme of the Forest Stewardship Council, an international nongovernmental organization (NGO) created to promote responsible forest management worldwide.

Case 3 comes from Honduras where new laws have moved legal ownership of large areas of the forests from the national government to municipalities. However, local people found this did not solve many of their problems, so they have organized cooperatives and federations of cooperatives to develop the power to negotiate better contracts, access and marketing.

A descriptive analysis of IFRI data about the main characteristics of forests in South America is presented in Section 5 in order to shed light on the opportunities and challenges that local forest-user groups face there. In this section, we also introduce our expectation that we will find healthier forest at sites where user groups have reached more cohesive levels of organization in their collective activities.

Also within this section, descriptive statistics and a simple cross-tab analysis are used to examine empirical data on attributes and characteristics of the settlements or villages, the forest-user groups and the forests themselves. Researchers in the IFRI research programme collected this information, visiting 34 sites in Bolivia, Ecuador, Guatemala, Honduras and Mexico between 1994 and 2002. Information from each of these sites was collected following the same research protocol and methods, making the data comparable.

The descriptive statistics generated from these sites illustrate the variety of local conditions that exist in our sample. The cross-tab analysis assesses the relationship of the local institutions with the forest conditions for each of the sites by testing whether there is a statistically significant association between highly organized collective activities and relatively good forest conditions. Our results indicate that the stronger the local institutions for harvesting and monitoring the better the forest conditions, but we cannot conclude that strong local institutions are sufficient to solve every kind of collective dilemma.

2. The nature of forest goods

As human populations and their demands on forest resources grow, citizens and officials from around the world search for solutions to the problems of forest degradation and deforestation. Many factors contribute to make forests very challenging to govern effectively. Most of these challenges emerge from the biophysical characteristics of forest resources.

Policy scholars classify resources as public or private goods based upon two dimensions: (i) the ease with which potential outside users can be excluded from access to the resource = the 'excludability' of the good, and (ii) whether or not a good once consumed is available for others to consume = the 'consumption' of a good (Ostrom and Ostrom 1999). Consumption can be either subtractive or joint: the former being a good that once consumed by one household is not available to another household (e.g., a tree), and the latter depicting a situation in which a household can enjoy the benefit of a good without subtracting from the amount available to another household (e.g., clean air; Varughese 1999).

Therefore, goods vary in their underlying attributes. Goods that are nonexcludable and subject to rivalry, like fish in the ocean, are called CPRs. Public goods such as a stable climate are nonexcludable and nonsubtractable and benefit all human beings; while private

Table 1. Grouping of forest products based on two series of attributes. Source: Adapted from Varughese (1999).

		Consumption	
		Subtractive	Joint
Exclusion	Difficult	<i>Common-pool resources</i> Fodder, wood, herbs, fruits, water	<i>Public</i> Habitat for plant and animal species, local climate, watershed protection, carbon sequestration, reservoirs of biodiversity
	Easy	<i>Private</i> Fodder, wood, herbs, fruits, water	<i>Toll goods</i> Areas or products of religious significance

goods are both excludable and subtractable. Toll goods, such as toll roads, are excludable but nonsubtractable. Table 1 illustrates the complexity of forest resources by summarizing their different biophysical attributes.

The definition of common property varies among scholars. However, most definitions of common property rights include these elements: (i) a well-defined group of co-owners, who (ii) develop and adhere to a well-defined management regime, that includes (iii) proscribed access by owners and exclusion of nonowners, and (iv) rights and duties of owners with regards to rates of use of the common property resource (Feeny *et al.* 1990; Swaney 1990; Bromley 1991).

CPRs are all subtractable in their natural environment with poor excludability, and include resources such as fodder, wood, herbs, fruits and water. But once subtracted by a household they are converted into private goods and are therefore easily excludable. Public goods such as clean air and water can be consumed not only by the households sharing the forests but also by other people outside the forest. However, forests can also be referred to as toll goods if there are sacred areas where only selected persons are authorized to go, therefore excluding all other households by internal rules of the forest-user groups.

Market mechanisms are thought to be the best governance institutions for private and toll goods, especially because of their characteristic of easy excludability. Because of their nonsubtractability, public goods are generally thought to be best managed within the purview of government. CPRs, on the other hand, combine problematic aspects of both dimensions. Since they are subtractable like private goods, they can be overused or even destroyed, but since it can be costly to control access to them (= excludability), it can be difficult to restrict the rate at which they are consumed. Most natural resources that everyone cares about are common-pool goods. Examples include forest products, water for irrigation, and fisheries. Their effective management remains one of the most difficult tasks facing modern public policy. Most forest ecosystems share the principal characteristics of CPRs and, as such, they pose different and arguably more difficult challenges to governance than do smaller-scale resources, whether these be private or public.

Another complication in the use of forest resources is that certain types of forest use can produce significant secondary or external effects (= externalities). For example, harvesting trees on a hillside may alter the local species composition, the carbon uptake capacity of a larger forest ecosystem or, through silting, the quality of nearby streams and lakes. As a consequence, the effective governance of forest resources must take a broader view of management-related externalities in order to avoid negative effects.

Finally, the spatial and temporal nature of forests and their potential externalities often do not correspond with existing political jurisdictions. Forest borders are rarely equivalent

to political boundaries, giving local managers only a partial understanding of the total forest resource. It often takes decades before degraded forests can regenerate, and it can take just as long to understand the true impact of forestry policies. However, political institutions generally encourage officials to operate within shorter time frames. Thus, it is not uncommon for policy-makers to ignore some of the more serious forest-related problems.

In summary, forests exhibit many characteristics that defy simple policy solutions. CPRs, where the exclusion of potential users is difficult, can present management challenges to policy-makers and those that enforce policy. Forests possess important externalities with regard to atmospheric, hydrological and biological services, many of which are hard to quantify and control. Forests are also complex in the sense that they can generate a myriad of products such as wood for construction or fuel, wildlife that is trapped or hunted, and leaves, fruits, fodder, seeds, straw, shade, recreation, stones and fertile soil, along with scores of other products consumed by humans. All of these products can mature at different rates, may be managed using both consumptive and nonconsumptive approaches, and can possess characteristics of common-pool, private or public goods, all the while providing ecosystem services for localities, regions or countries.

2.1 The characteristics of common property forests

One of the most common sources of confusion about the management of CPRs is how to distinguish their characteristics from their associated property rights regime. Part of the confusion is likely to stem from the terminology itself. For instance, rights to a CPR such as a forest can belong to the general public (= state property), to a government (= governmental property), a private individual (= individual private property), or a group of individuals (= group-owned private property). Likewise, “when they are owned by no-one or paradoxically by ‘everyone’, they are used as open-access resources by whoever can gain access” (Ostrom 2003).

McKean (2000) shows that individual private property rights often do not provide the best basis for effective forest management. Her argument is that privatization of forests often leads to forest fragmentation, which may seriously disturb the proper functioning of the forest ecosystem. Because of these drawbacks, she suggests that forests are better suited for management under common property regimes in which larger chunks of contiguous forest have a higher probability of being maintained. Such systems also tend to be more efficient to administer.

The governance of forests as common property may have many advantages over treating them as individual private property, but that is not to say that it is free from its own problems. Forest users may have conflicting interests and goals with regards to their forest use. Furthermore, individuals within the same forest-user group may have different levels of knowledge about the resource and different access to information, economic resources and political power. These asymmetries complicate the individuals’ efforts to achieve successful joint outcomes, i.e., the way in which they should manage their collectively owned forest. Social scientists have given considerable attention to the problems that challenge the governance of forests as common property. The next section reviews the main findings of this vein of social science research.

2.2 Challenges in governing forests as common property

Collective-action problems “occur when individuals, as part of a group, select strategies generating outcomes that are suboptimal from the perspective of the group” (Ostrom and Walker 1997). These problems are caused by a lack of information, difficulties in coordination, the existence of obstacles to exclusion and rivalry of extraction (Poteete and Ostrom 2002). This leads to information asymmetries, and motivational problems then arise (Ostrom *et al.* 1993). Any or all of these can lead to the deterioration of a community’s forest(s). Some communities are able to overcome such problems by developing highly organized institutions to deal with them, while others are not.

Using a range of field observations, Ostrom (1990) showed that it is possible for local communities to self-organize in ways that resolve complex collective-action problems related to natural resource governance. In fact, local communities have often demonstrated that their self-organized efforts in natural resource management can outperform government programmes (McCay and Acheson 1990; Ostrom 1990; Feeny *et al.* 1998).

Nevertheless, institutions that effectively manage forests do not have to be self-organized, as it is possible for positive collective outcomes to occur through government coercion. This was the accepted approach to forest management as recently as 30 years ago in several areas of the world (Arnold 1992, 1998; Wunsch and Olowu 1995). The policies that flowed from coercive control emphasized the 'scientific' exploitation of forests within a context of economic return (Richards and Tucker 1988; Scott 1998). We now know that such an approach can be ineffective in forest management, not only because of the costs involved, but because local communities will often simply not buy into the centralized state programme. In regions of resource scarcity, where forest products are essential for local livelihoods or where forests are isolated, state-controlled agents are commonly unable to secure, manage or protect the forests. Such a top-down strategy can also generate undesirable side-effects, for example, policies may not be flexible enough to cope with local ecological variation or local people's needs, knowledge and preferences.

Significant shifts in accepted forest governance standards have occurred in the last 30 years or so. One of the major changes has been to confer property rights over forests to local communities. Based on the idea that local communities live with forests, are primary users of forest products, and often create *de facto* rules that significantly affect forest conditions, scholars and policy-makers argue that more equitable and effective outcomes can frequently (though not always) be reached by transferring *de jure* rights over forests to local communities (Perry and Dixon 1986; Arnold 1990; Bhatt 1990; Dei 1992; Douglass 1992; Ghai 1993; Raju *et al.* 1993; Ascher 1995; Clugston and Rogers 1995; for reviews, see Wiesner 1990; Baland and Platteau 1996). The core theory is that local users hold important time- and place-specific knowledge necessary for the creation of successful forest management, particularly for the institutional arrangements that are needed to achieve success (McCay and Acheson 1987; Berkes 1989; Ostrom 1990, 1992a; Bromley *et al.* 1992; McKean 1992; Peters 1994; Wade 1994).

With community management of forests now receiving increased attention from policy-makers, an empirical literature is developing that addresses the causes of success and failure for community-based natural resource management. In her 1999 paper, Ostrom suggests that although the necessary conditions for effective communal forest management vary across contexts and countries, there are some that are basic to good local governance. These conditions can be grouped into two sets of variables: the attributes of the resource and the attributes of the users, as described in Box 1 (overleaf).

If users have the attributes listed in Box 1, they are more likely to be willing to invest time and effort in the management of their forest resources, unless the resources are perceived to be threatened in some way and competition for them has become acute. A positive trade-off between costs and benefits must exist in order to achieve effective communal forest governance. Provided there is sufficient information available, forests in good condition with enough available products will give individuals more choice. Under these conditions, individuals are more likely to develop practices that can produce and sustain a self-organized system of forest management.

The resource attributes described in Box 1 help to define scenarios where organizing for communal management pays off. However, this will only be the case if the forest-related products generate significant monetary or subsistence income for local populations, and possible sources of conflict are minimized so that agreements can be made based on trust. Among the attributes of the forest, size is a particularly important determinant of the success of communal forest management. Another important contextual factor is the autonomy that allows forest-using individuals to craft forest management rules themselves and to implement them.

Box 1. The conditions for successful communal forest management

Attributes of the resource

Improvement potential

The resource is not either so degraded or so underutilized that attempts to organize it are pointless.

Indicators

Reliable and valid information about the state of the resource is available at a reasonable cost.

Predictability

It is relatively easy to predict the supply of forest products.

Spatial extent

The resource is small enough so that users, given the transportation and communication technology available, can understand its microenvironments and know where its boundaries are.

Attributes of the users

Salience

Users are dependent on the resource for a major portion of their livelihood or subsistence.

Common understanding

Users have a shared understanding of the resource and of how their actions affect it and each other.

Discount rate

Users set a sufficiently high value on the future benefits to be derived from the forest resource to make community management appear attractive.

Distribution of interests

Users are equally affected by coordinated management of the resource, irrespective of power or wealth.

Trust

Users trust each other to keep promises and to form mutually beneficial relationships.

Autonomy

Users are able to determine access and harvesting rules without external authorities countermanning them.

Prior organizational experience

Users have learned at least minimal organizational skills through participation in other local associations or from neighbouring groups.

Source: Ostrom 1999.

While this list is by no means an attempt to exhaust all possible determinants that shape self-organized management of common property forests, it does provide a comprehensive account of some of the more influential and relevant ones. However, one should keep in mind that, for any given situation, this set will depend upon the environmental and socioeconomic contexts.

3. Communal forest management in South America

This section provides a brief historical overview of some of the features of communal forest management that are particular to South America. Compared with other developing regions, including East and West Africa and South Asia (Agrawal and Ribot 2000), South American forest-user groups have access to relatively valuable forest resources. While the legislation to enable community forestry in South America is fairly recent, the challenge for rural communities to govern their forests is not new. Historically, community forestry activities have been important to the livelihoods of most rural people in South America, even though formal state governments have not always been supportive of such activities.

Most of the independent South American republics gained their freedom during the 19th century. Since then, these newly created countries have continued to follow the old colonial rulers' policies of confiscating large land areas from indigenous people, allowing them to keep only small areas of forestlands in comparison to what they originally possessed. Beginning in the 1950s, many developing countries nationalized such natural resources as land and forests to improve forest management. These areas were converted to *de jure* government-property regimes, eventually to become *de facto* open-access regimes (Arnold 1998). Governments divided forested CPRs into individual tracts, and in due course these lands were then distributed as private property to members of the ruling elite, and not as government concessions as had been the colonial tradition. While such 'reforms' improved the security of forest owners' tenures, they also had several undesirable socioeconomic consequences, such as skewed distribution of forest property rights. These privatization reforms offered lucrative opportunities for the richer members of society, while poorer groups lost access to the resources on which their livelihoods depended.

By the mid-20th century, most South American countries had ongoing government programmes promoting mass colonization of both previously undisturbed tropical forests and forests inhabited by indigenous populations. Policies like these formed integral parts of governmental strategies to relieve pressure on increasingly scarce agricultural lands in more heavily populated regions and to stimulate growth of export-oriented agricultural economies.

Thus, even by as late as the mid-20th century, conservation of forest resources was not an issue for the South American governments, nor were they concerned with the customary rights of indigenous peoples. Typical government policies and practices continued to remove indigenous populations from their lands or severely reduce the geographical extent of traditional lands. Policies were aimed at improving the operating and ownership conditions of large-scale, private agricultural and timber entrepreneurs, although uneven implementation of policies led to situations where forests might either be publicly owned by the nation-state, *de facto* owned by private citizens, granted to private citizens under formal private property rights transfers, or remain open-access resources. A national policy bias towards agriculture, along with land-use conflicts among users with overlapping claims, produced a high level of forest tenure insecurity. As a result, rather than stimulating the investment of time and money to produce longer-term management systems and sustainable yields, maximization of short-term profits dominated the logic of forest management practices in the region (Pacheco 1999; Contreras and Vargas 2001; Andersson 2002).

Despite the fact that most South American countries have at one time or another reformed land and forest legislation in favour of private rights, various forms of collective access to natural resources have persisted. These are exemplified by community institutions in Bolivia, Ecuador, and Peru, 'ejidos' in Mexico and Guatemala, and indigenous territories in Bolivia and Colombia. Therefore, most of the recent legislative changes at the constitutional level have had no practical implications for many local people who were already managing the resources as if they owned them.

By the end of the 20th century, most South American countries had introduced a new set of policies that aimed to encourage conservation of the environment and to restore

indigenous groups' rights to land. The change in some countries is occurring through a two-fold process whereby NGOs, which can be local, regional or international, help indigenous groups to articulate their demands, while central government organize the top-down transference of rights (Hernáiz and Pacheco 2001; Urioste and Pacheco 2001).

Today, the national governments of most South American countries publicly acknowledge the importance of local institutions in sustainable natural resource management, although much of this has yet to be implemented. While some recent policies have supported *de jure* rights of previously ignored local forest-user groups and of local (municipal) governments, it is only now that they are beginning to translate into real empowerment of community-level institutions in forestry sector governance. Interestingly, recent studies in Bolivia, Colombia and Guatemala have shown that those newly empowered municipal governments that emphasize collaboration with a variety of governmental and nongovernmental actors, including local forest-user groups, perform far better in supporting forest tenure security than municipalities that follow a classical top-down governance scheme (Gibson and Lehoucq 2003; Andersson 2004).

4. Opportunities and challenges of community forestry

In this section, we introduce the opportunities and challenges that local forest-user groups face in managing their forests in South America. We begin by presenting three community forestry case studies from Bolivia, Mexico (from Tucker *et al.* 1999) and Honduras to illustrate how communities can organize their forest use in order to take advantage of newly occurring opportunities.

The case studies are analyzed on the premise that the performance of local institutions is crucial not only to successful communal management of forests but to sustainable forestry practices in general. We then further examine our hypothesis in relation to data that were collected during fieldwork by IFRI's Collaborative Research Centers (CRCs) in Bolivia, Ecuador, Guatemala, Honduras, and Mexico between 1994 and 2002. We include a set of attributes and characteristics from 34 settlements and 47 forest-user groups who have access to forest areas of variable size. These were selected to reflect the heterogeneity of the South American region in terms of rural communities' relationships with forests.

A forest-user group is defined as the people who share the same rights and duties to products from the forest(s) (e.g., a group might be devoted to timber harvesting, gathering of medicinal plants or fruits, and hunting), even though they may or may not be formally organized. A settlement refers to a local jurisdiction inhabited by one or more forest-user groups. It could also be referred to as either a community or a rural village. There is not a one-to-one relationship between the number of settlements and forests, since a settlement can encompass more than one forest; therefore the number of forests can be larger than the settlements, as in this study.

Case 1 – Forestry management in the Yuracaré territory, Bolivia

The Yuracaré are one of Bolivia's indigenous groups. In the past, the Yuracaré have been quite mobile within large territories, but during the 19th century they settled within the Chapare River watershed. This territory is now home to 11 Yuracaré extended families, each composed of 11 to 20 nuclear families. According to the 2001 census, the population of the entire group was then 2358 people.

During the last 20 years, Bolivian forestry regulations have created conditions under which indigenous groups can commercially exploit the forest. As a result, the Yuracaré have become motivated to organize local forestry associations for commercial timber exploitation, and they have adapted their social organization and forest management practices to the scope of the regulations. Like the other indigenous societies that live in Bolivia's lowlands, the Yuracaré traditionally rely on self-governing institutions to manage their natural resources.

By 1992, the Yuracaré had created a forestry association and had received an annual permit from the national forestry agency (CDF, Centre for the Development of Forestry), authorizing a regulated timber harvest. Within the association, each Yuracaré negotiated the amount of timber to be harvested individually according to their own experience and needs. This practice reinforced each Yuracaré's knowledge of the territory as a whole, and did not compromise the ecological conditions of the forests. There were no technical rules for cutting trees in terms of size, rather the Yuracaré developed their own system of forest classification, and harvesting occurred in relation to maturity. The Yuracaré harvest rotated within their territory, not because of a lack of resources but owing to their perception of 'using without depleting'. Normative social behaviour was an important mechanism that controlled the relationship between forest use, harvest and ecological conditions, and this permitted the Yuracaré to secure both forest and societal sustainability over time.

In 1996, new land regulations in Bolivia transferred ownership of nearly 20% of the country's lowlands (20 million ha) to about 30 indigenous groups in the form of common property. The Bolivian lowlands encompass roughly 70% of the country, and approximately 80% of Bolivian forests are found there. The territories of the indigenous groups were recognized along with rights of customary ownership of renewable resources and governance. In 2000, the Yuracaré were given title to 240 000 hectares.

This new regulation launched a process that reinforced the Yuracaré forest management system within a timber production paradigm, but it also required greater regulatory detail such as the formulation of a Forest Management Plan for the collective management of the forests. Since timber production was traditionally an individual undertaking, the Yuracaré were now required to develop new institutions to deal with the opportunities of the new legislation. They began a process of learning through experiencing. The Yuracaré are currently facing some important challenges such as how to bring together collective and individual management, how to harmonize collective and technical decisions, and how to arrange joint forest management contracts with business enterprises without losing institutional capacity.

Today, the Yuracaré control the entire Chapare River watershed, practising farming, hunting, fishing and gathering of forest resources, mainly within a context of household self-consumption. The population is strongly dependent on the forest as the main source of food for its survival. Activities include itinerant seasonal agriculture, hunting, fruit and seed collecting for food and medicine, and harvesting of both construction materials for dwellings and timber products for the marketplace. Part of their economy is based on forest products and is thus subject to internal and external market dynamics (Becker and Leon 2000).

Case 2 – Forest management in Capulálpam, Oaxaca, Mexico

It is estimated that as much as 80% of Mexico's forests can be divided into two common property categories, 'ejidos' and indigenous community lands. The 'ejido' is a traditional management system for land and natural resources dating from precolonial times that establishes common property rights for small-scale landholders and individuals with usufruct. Each 'ejido' encompasses a number of landholder households. Both 'ejidos' and indigenous community lands are governed and used by those who have rights under customary rules, hence self-organization is known in this part of the world.

The municipality of Capulálpam de Méndez (Capulálpam) is a Zapotec-speaking indigenous community in the State of Oaxaca, Mexico. Capulálpam is located in the Sierra Norte to the northeast of Oaxaca de Juárez. It has a complex topography: the municipality is composed of 315 families whose dwellings are dispersed across a sloping pine-oak forest landscape and surrounded by agricultural fields. Community inhabitants practise many long-standing Zapotec traditions.

In the 1960s, a conflict arose between the indigenous community and the state government which had granted a large timber company a forest concession. Because the inhabitants of Capulálpam were not benefiting from this commercial arrangement, they organized a major protest, claiming rights to the forest concession. The campaign was

successful and, in the 1980s, what has started as a protest movement became UZACHI, a Zapoteco–Chinanteca union that encompasses people from these indigenous Mexican societies, with duties to plan and monitor the management and use of the forests.

Today, the community enjoys exclusive ownership rights to the forests. It has received a forest certificate from the Forest Stewardship Council and a government-approved Forest Management Plan guides its decision-making. According to this plan, the forest has been separated into management units so that intensive use is limited to specific areas of the forest for given periods of time. The forest management plan is supervised with the help of UZACHI technicians. However, the assembly of *comuneros*, composed of all the adult community members, determines most of the policies for the administration and use of the forest resources, as well as the administration of the communal sawmill.

Income from timber sales is invested back into the community, mostly for improvement of the municipal infrastructure. The benefits of a communally operated forestry firm are local jobs and further contributions to the municipal budget. While income from commercial forestry is not sufficient to meet all of the basic infrastructure needs of the small municipality, it does represent a crucial source of income, and it has helped to stimulate thinking and action in local forest conservation.

Case 3 – Community forestry in Honduras

Honduras is a country where 80% of the land slopes on grades of more than 15%, substantially limiting the areas where agriculture can be practised. Nevertheless, Honduras is rich in forests, with approximately 50% of the country forested. Many of the people living in and around forests see communal forestry as a way to improve their livelihoods, and many of them are organizing themselves to take advantage of new legislation that has moved legal control of large forest tracts from national administrations to municipal governments. Honduras is an interesting example of government-supported institutional forestry reform and of local people trying to address forestry-related problems by forming cooperatives.

There are more than 80 locally formed forest product cooperatives in Honduras, the majority of which work in the area of resin collection. Other cooperatives deal with timber, including a few that have evolved from manual milling to more sophisticated techniques. For example, in the country, the market control of resin has progressively been restricted to three firms, mostly because of the low prices that the large resin-collecting companies can offer. This limits the opportunities for local producers who cannot compete with the large companies. To overcome these drawbacks, the cooperatives have amalgamated into federations in order to deal more effectively with common problems, with the objective of organizing transport to sell resin in neighbouring countries for better returns.

Recently, local community leaders, managers from forest-related cooperatives, and officials from the Social Forestry division of the Honduran Forest Service met in the small community of Yamaranguila to discuss forestry issues. Some of the problems raised by community leaders at this meeting were the difficulties in finding resources to cover the costs of producing management plans (which must be done before timber is harvested) and the lack of land ownership that limits access to forests. The community leaders addressed the benefits of a potential new forestry law that would help clarify these issues.

Forests currently managed by municipalities require that municipal mayors sign and forward petitions to the national government before local use can occur. Although mayors are locally elected, in many cases they are nominated by political parties whose headquarters are located in the Honduran capital Tegucigalpa, and they spend most of their time there. While some community leaders reported that their mayors were helpful in preparing and forwarding forest-use requests, others reported that their mayors were frequently absent and did not seem interested in advancing the paperwork required for local access to resources or for transport of lumber. The most helpful mayors were found in the indigenous Lenca municipalities where forest protection and environmental

movements have been organized for more than 20 years. The bureaucratic paperwork would be eliminated if cooperatives were granted access to the forests as owners.

Representatives of local cooperatives and government officials are now encouraging reform legislation with the objective of clarifying the norms that regulate local access to forests. The reform is also aimed at increasing local control over forest access, developing less costly management plans and promoting management of forest resources with the participation of local communities.

5. Attributes characterizing community forestry

The IFRI Research Programme is a multicountry programme that functions through an international network of Collaborative Research Centers (CRCs) in South America, Asia and Africa. IFRI carries out field research by gathering comparable data about the relationships between rural communities and their forest resources (IFRI 2002). Each CRC uses multidisciplinary teams and the same ten research protocols to collect data on the biophysical conditions of forests at designated sites, and on related socioeconomic, demographic, cultural and institutional variables. Techniques such as participatory rural appraisal, surveys and interviews with key informants are used both individually and during group meetings.

The purpose of this section is to examine the role of local institutions in forest management. Because local forest users seldom take national forest legislation at face value, that is, they will not automatically obey rules without considering their effects on either their individual or collective interests, forest management by user groups and its effect on forest conditions depends to a large extent on how forest users organize themselves. The local rules or institutions for forest management determine the effectiveness of such organizational efforts by constraining or rewarding specific types of behaviour among user-group members. Consequently, local institutions play a mediating role in forest governance. In this section of the chapter we will test the hypothesis that local efforts by forest-user groups to collectively organize forest harvesting, marketing, and monitoring are positively related to superior forest health.

The analysis occurs in two parts. In the first part we propose conditions under which collective activities are expected to occur by looking at local variation in several attributes of the resource and of the forest users. Our purpose is to provide contextual background for the 34 sites that we consider in the analysis (Table 2, overleaf). The descriptive statistics that we present illustrate the variety of local conditions that exist in our sample. In the second part, we analyze relationships among the local institutions and existing forest conditions at selected sites. We carry out a simple cross-tab analysis to test whether there is a statistically significant association between strong local institutions and good forest conditions.

5.1 Local conditions for communal forest management

Our analyses are based on attributes that were directly measurable through IFRI field research, such as property ownership type, degree of forest deterioration, predictability of forest products, size of forest, salience of forests, and the culture of cooperation among local users.

Earlier research (Ostrom 1992b; Gibson *et al.* 2000) showed the importance of **property rights** in excluding outsiders from forest exploitation and thereby for providing incentives to forest stakeholders to make forest improvements. While South American forests have differences in ownership and/or rights regimes – and some governments have conferred significant control of forest lands to local communities during the last decade – most countries have retained ownership of large tracts of forest resources under national or local governmental control.

Figure 1 (on p. 41) shows that about 50% of the sampled forest lands were in the public domain, owned by national and local governments, with local governments holding about 30% of the total lands. The other half of the forest lands were privately owned: 42% being

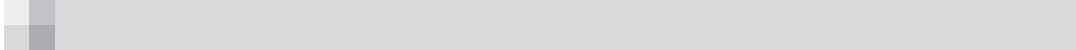


Table 2. List of the settlements, forest-user groups and forests sampled in this study.

Country	Settlement N = 34	User group N = 47	Forests N = 36
Bolivia	Huayco Grande	Rinconehos	El Huayco
	San Antonio de Juntas	User group of Belén Cruz	Belén Cruz
	Villa Aquiles	Property owners and tourists	Villa Aquiles
	Lagunillas	Property owners in Lagunillas	Lagunillas
	San Juancito	San Juancito community members	San Juancito
	San Lorenzoma	San Lorenzoma community	San Lorenzoma
	La Merced	Community La Merced	La Merced
	Corregimiento of Trinidadcito	Logging enterprise, local community and loggers	Trinidadcito
	Cachimayu	Settlers of Chapis	Chapis Cachimayu
	El Saire	Herders of Chapis	Chapis El Saire
	Huacanqui–Padacaya	Owners of Chapis	Chapis Huacanqui
	La Merced, Totorá, Río Negro, Ibare, Nueva Galilea	Settlers	La Merced, Totorá, Río Negro, Ibare, Nueva Galilea
	San Antonio de Misiones	Community and timber enterprises	San Antonio
	Santa Anita	Community of Santa Anita, livestock owners and loggers	Santa Anita
Guatemala	Morán-Naranjo	Community of Morán-Naranjo	Morán-Naranjo
	Community of Las Cebollas	Las Cebollas community members	Las Cebollas
	Finca	Community of San Jose	San José
	San Jose	Community of San Jose	State of San Jose of Minas
	Socorro	Community of Socorro	Forest of Socorro
	Tesoro Community	Tesoro	Tesoro community forest
	Finca Dulce Nombre	Tachoche	State of Tachoche forest
Honduras	La Campa Centro	Center of La Campa, cooperative of La Campa, independent potters and Lacamperos	Bosque Comunal de la Campa
Mexico	Capulalpam	Community of Capulalpam, and mushroom gatherers	Capulalpam
	Cerro Prieto	Ejidatarios' wives, ejidatarios, avenos, ejidatarios' sons and families	Cerro Prieto absorption area
		Tourists	Cerro Prieto core zone
	Donaciano Ojeda	Irrigation committee 3rd Manzana, 2nd Manzana, 1st Manzana, community and families, and residents	Donaciano Ojeda
	Huayapam Centro	New residents and traditional settlers	Huayapam
	La Ponga	Invaders	Protective forest Loma Alta
	La Union, Loma Alta, Suspiro	Land owners in protective forest	La Union, Loma Alta, Suspiro

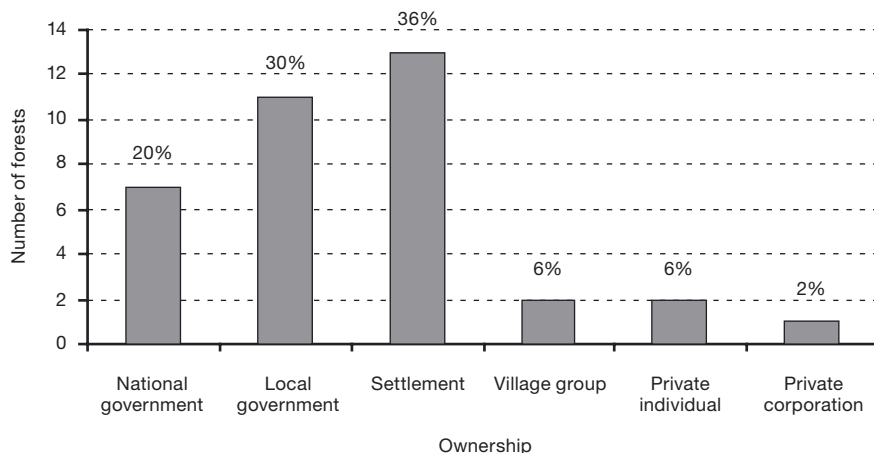


Figure 1. Breakdown of forest ownership in the 36 selected forests examined in this study (see Table 2).

held in common as property of the settlement or of a village group, with the remaining 8% held as private property belonging either to individuals or corporations. In trying to understand the conditions that would favour collective action, it seems reasonable to assert that those communities whose members hold forest rights in common would be more likely to develop institutions to manage the forests than would communities or individuals that use forests on government-owned lands. This is because the less secure the property rights are, the less certain are the prospects of recovering the resources that are managed or produced through collective activities.

Forest **deterioration** is another variable that helps to explain the motivation of forest-user groups in managing their forests. In order to come up with a better appreciation of the ways that forests can deteriorate (or lose value), four IFRI forest attributes were selected: vegetation density, species diversity, commercial value and subsistence value.

Based on evaluations by botanical experts, Figure 2 (overleaf) shows that the conditions of the case study forests vary a great deal among the samples, and that the values associated with forest conditions are more or less normally distributed. Most expert opinions fall into the 'about normal' category. The attributes of roughly one-fifth of all forests assessed were considered to be in poor condition (very sparse or somewhat sparse), and a slightly higher proportion were considered to be in good condition (somewhat abundant or very abundant).

Expert botanists used rapid visual assessments to determine forest vegetation density and species diversity. The commercial and subsistence values of forest resources were analyzed using data on the principal forest products (whether used for subsistence or trade), collected according to IFRI's protocol. The following possibilities were offered to assessors for each variable: very sparse, somewhat sparse, about normal for this ecological zone, somewhat abundant, and very abundant.

We expected to see better conditions, that is less deterioration, in forests managed by user communities with a higher level of institutional development. This is because communities that have developed ways to work together effectively will generally do a better job of controlling access to a resource, thereby reducing the adverse effects of competition. A forest in good condition can also be an important stimulant for continued institutional development. People who do not have access to a forest in good condition may not have enough at stake (= a cost-benefit assessment) to be motivated to self-organize and assume management responsibilities for the resource.

The **predictability** of forest products, that is, their availability from season to season, is also important for the cost-benefit calculations that forest users undertake when making

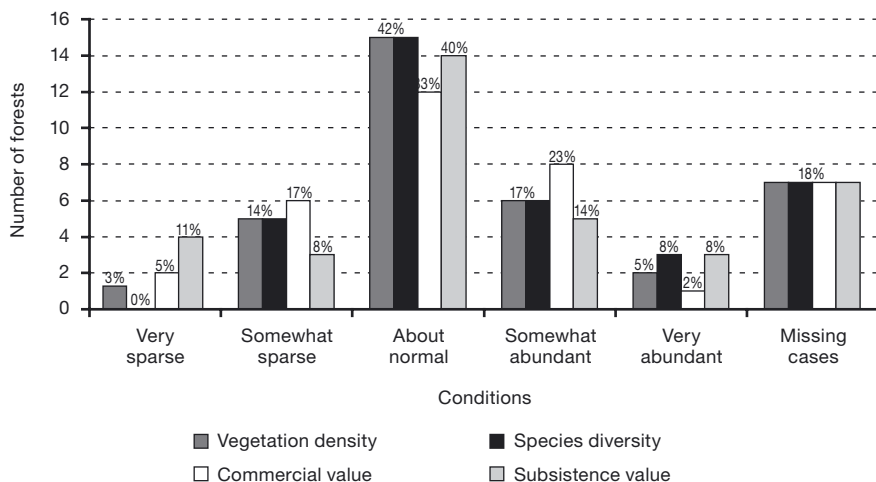


Figure 2. Assessment of forest conditions in 36 forests examined in this study (7 missing cases) developed by expert botanist members of IFRI teams. Six categories are used to define different forest conditions, based on the assessment of tree density and floristic richness.

decisions about whether to engage in collective management activities. The temporal variability of resources is recorded in Figure 3, and we note that more than half (53%) of forest products are perceived as not varying from season to season. Fifteen percent of products are viewed as having little or moderate temporal variation in availability, and products varying substantially and dramatically only reach 10% and 8% respectively. Fourteen percent of the forest products lack this data and are considered missing cases.

Our expectation was that communities facing higher levels of uncertainty in forest product predictability would be less motivated to organize and establish the rules for collective actions required to manage a forest and its resources. Such uncertainties are more likely to discount the perceived net benefits to be gained from organizing.

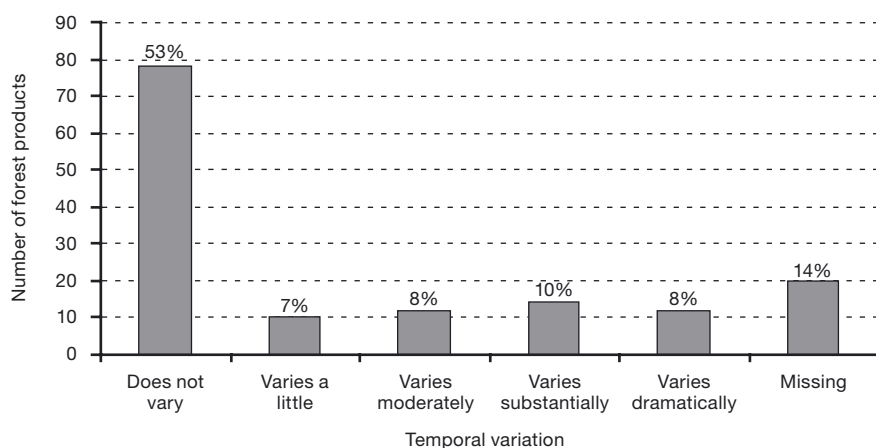


Figure 3. Predictability of future (next year) availability of 146 forest products in 36 forests (20 missing records). Temporal variation is described using six categories.

The **size** of forests is one of the more influential contextual variables that motivates community self-organization in forest management. We found that 52% of the sampled forests were less than 500 hectares, 28% fell in the range of 501 to 3000 hectares, and 20% were more than 3000 hectares (Figure 4). Forests managed by forest-user groups are relatively small: 89% of the forests are less than 5000 hectares.

Nevertheless, the effect of forest size on the development of community forest management is not a direct relationship. For example, those communities with access to larger forests can have difficulty in establishing regulatory bodies to limit access to outsiders, monitor extraction of products by outsiders, and monitor use of forest resources by members of the community entitled to them. Communities managing larger forests will also need more time and effort to develop institutions and to undertake actions that will result in better forest conditions. Furthermore, the capacity of a local user group to manage a larger forest will also be important. This can sometimes be evaluated by examining the ratio of forest size to the number of user-group members; there is usually an upper threshold of how much forest a given user group can effectively manage.

When analyzing **salience** of forest products for the people using them, we first asked how many households in the forest-user groups depend significantly on the forest for their own subsistence, which, on average, was 63% of the people interviewed. Secondly, we asked how community members derived their basic income. Combined responses showed that 63% of the people interviewed in the settlements derive their basic livelihoods from subsistence farming, 9% from commercial farming, and 9% through waged labour. Those people in the 'other' category (18%) gained their income mainly from artisanship and animal husbandry (Figure 5, overleaf).

Harvesting forest products was not ever given as the main source of income. As a result, we concluded that forestry activities are important for family subsistence but are a complementary source of income, secondary to income from agriculture and paid manual labour. Therefore, forest users' main source of livelihood is a combination of agriculture for both subsistence and cash income, with gathering of forest products and hunting game for subsistence. The impact of forest users on the forests is thus complex. On the one hand, there is a need to clear old-growth forests for agriculture but, on the other hand, there is also a need to conserve forests for subsistence products.

Members of some communities will fail to create and enforce rules to counteract over harvesting, while others will succeed in creating and enforcing rules that constrain

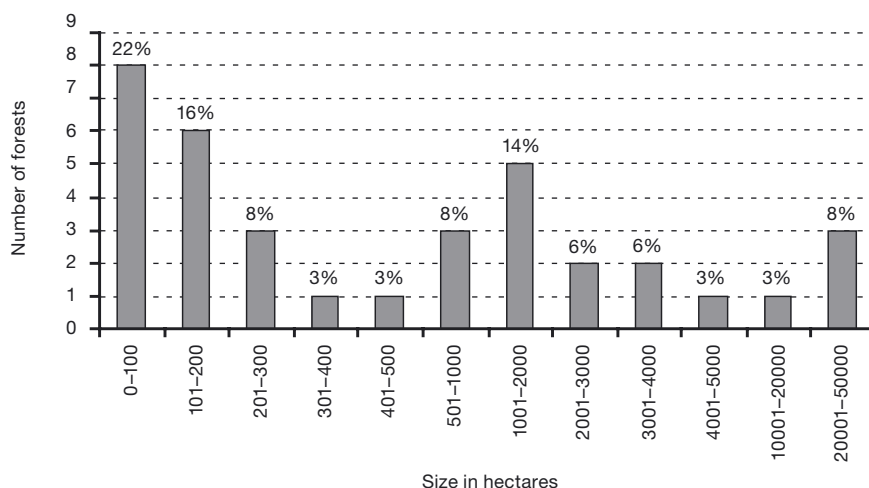


Figure 4. Assessment of the size of the forests (ha), across the 36 forest sites included in this study.

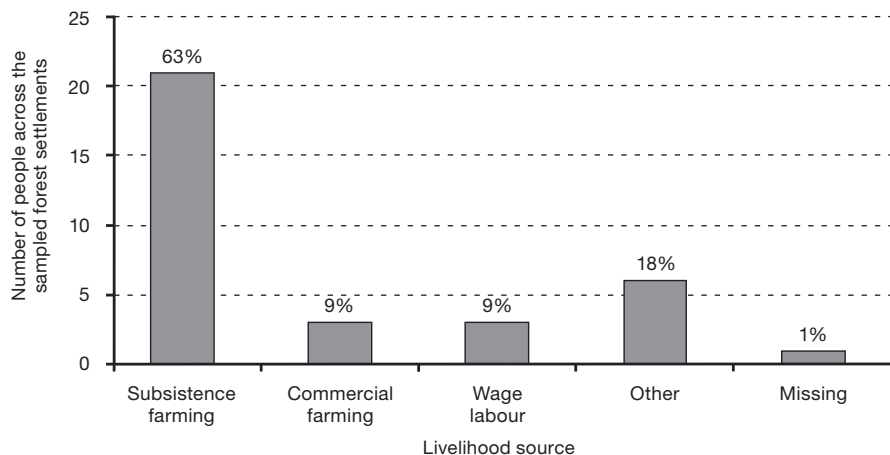


Figure 5. Assessment of rural livelihoods in 34 settlements (1 missing record). Four categories are identified. Other = income mainly derived from artisanship and animal husbandry.

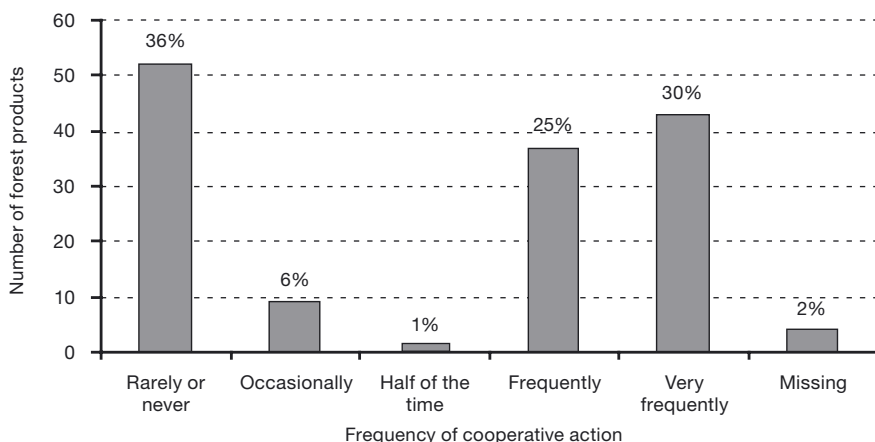


Figure 6. Assessment of culture of cooperation for the exploitation of 146 forest products (4 missing records). The 5 categories identified indicate the frequency of occurrence of cooperative actions with respect to the collection of a certain number of products.

resource-damaging individualistic behaviours and/or reward resource-enhancing behaviours that contribute to the collective good (Ostrom 1990). This success or failure is related to a community's **culture of cooperation**. Our assessment (Figure 6) focuses on rules in-use rather than rules in-form (e.g., national forest regulations). We partition the observed community responses into five categories ranging from 'rarely or never' cooperate to following the rules 'very frequently'. Our analysis was based on activities observed during the harvest of 146 forest products. For 36% of the forest products, user groups do not or rarely follow community-established rules in exploiting these resources, while for 55% of the forest products group members obey rules frequently or very frequently. The remaining 7% of the forest products fall in between these two extremes.

There was significant variation in the culture of cooperation among user groups. In our sample, those who engaged in more cooperative interactions with group members would be more willing to continue this behaviour as trust is developed and cooperation pays off. Those forest users that do not obey rules are less likely to develop trust and engage in the reciprocal behaviour that will, in turn, affect institutional development. We thus expect that user groups enjoying higher levels of trust would also cooperate more effectively when it came to forestry activities.

When we considered the combined average responses, we found that community forestry had developed under normal forest conditions in a variety of types of property regimes in relatively small forests where forests were important for subsistence, and where there was a greater culture of cooperation, and less so for commercial purposes, where there were no significant changes in the availability of products over time. We now address the issue of whether organizational efforts are associated with superior forest health.

5.2 Do local institutions matter?

If institutions influence the variables listed in Box 1, we would expect to see better conditions in forests where user groups are engaged in more highly organized collective activities. Here we use observations made of 47 IFRI forest-user groups to evaluate the significance of the relationship between their collective forestry activities and forest health, using a Chi-square test of the association between dichotomized variables as explained below. We chose the Pearson Chi-square test because it is well-known, easily executable in cross-tabs, and easy to interpret.

IFRI protocols query when and how often individuals in each user group interact, classifying the options as follows: year round, seasonally, occasionally and never. Among the categories of interaction specified in this ordinal variable, the most important for communal forestry management are cooperative harvesting, cooperative marketing, and cooperative monitoring/sanctioning. The frequency of interactions in such categories allows us to compute the degree of collective activity in the 47 forest-user groups. In order to simplify the analysis, we created two categories: 'high level of collective forest activities' and 'low level of collective forest activities'.

If the groups interacted only occasionally or if they never interacted we assigned them a 'low' level of collective forest activity and, conversely, if the groups interacted seasonally or year-round, we assigned them a value of 'high' collective forest activity.

We relied on the expert opinion of local foresters to assess forest health. At each site, we asked the expert to classify the forest according to an ordinal scale ranging from 1 (degraded) to 5 (very good). In order to simplify the analysis, we dichotomized this variable into 'good' vs. 'degraded' forest health, reducing the categories identified from five to just two (see Figure 2 and Table 3, overleaf). A forest is recorded as 'degraded' if forest conditions with regard to tree density and species diversity are described as very sparse or somewhat sparse, while a forest is categorized as 'good' if it is described as normal, somewhat abundant or very abundant.

The results of a cross-tab analysis are presented in Table 3. The results indicate that stronger local institutions for harvesting and monitoring forest resources are associated with superior forest health. This positive association is statistically significant at the 95% confidence level (Chi-square = 9.476, Df = 1, $P < 0.05$ for collective harvesting, and Chi-square = 4.102, Df = 1, $P < 0.05$ for collective monitoring).

Our results suggest further that the effects of local institutions are not the same for all aspects of forest management. For instance, we found no significant relationship between collective activities in marketing and superior forest health. Consequently, one cannot conclude that local institutions are sufficient to solve all kinds of collective problems. Another limitation of this analysis is that we cannot say what is enabling some user groups to organize their forest activities while others have not been able to do so. However, we do know that the sample includes a wide variety of local conditions, and it is not biased towards communities with a higher propensity for cooperation (Figure 6).

Our results are consistent with earlier findings in the literature about the importance of institutions as mediating or mitigating factors in forest management. For example, Agrawal

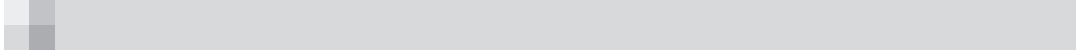


Table 3. Relationships between different types of collective activities of 47 forest-user groups and forest conditions. Df = degree of freedom, a measure of the precision of an estimate of variation.

Collective activities		Forest health		
		Degraded	Good	Total
Cooperative harvesting	Low	22	1	23
	High	10	8	18
Total		32	9	41
Chi-square test		Value	Df	Significance
		N=41 6 Missing cases	1	0.002
		Degraded	Good	Total
Cooperative marketing/ sales	Low	19	3	22
	High	12	6	18
Total		31	9	40
Chi-square test		Value	Df	Significance
		N=40 7 Missing cases	1	0.138
		Degraded	Good	Total
Cooperative monitoring/ sanctioning	Low	10	0	10
	High	22	10	32
Total		32	10	42
Chi-square test		Value	Df	Significance
		N=42 5 Missing cases	1	0.043

Df is equal to the sample size (2) minus the number of parameters estimated in the test, in this case 1.

and Yadama (1997) argue that institutions can reduce demographic and socioeconomic pressures on the forests of the villages of the Kuman region in India. Varughese (1999) studied the role of institutions that mitigated the dynamics of population change and group size to find that in 18 cases from Nepal population growth rates were not correlated with variations in forestry conditions, but collective action was. These same authors also concluded that ethnic, social and economic heterogeneity did not have a determining effect on either the likelihood or success of collective forest management.

The importance of institutions in various forest conditions was tested by Gibson *et al.* (2003) using more than 150 IFRI cases in 12 African, Asian and South American countries. They found consistency of rule enforcement to be more important than levels of cooperative forestry activities in explaining variations in the condition of community forest resources.

6. Conclusions

Forests are being degraded throughout the world and foresters, rural developers and environmentalists are searching for ways to reverse this trend and achieve sustainability of the biophysical landscape, biodiversity and rural livelihoods. In South America, many national governments have now concluded that local governments, communities

and individuals should become more active in managing forests. What types of local management arrangements are most appropriate in different contexts? This chapter has shown that when analyzing local forest management systems it is important to look beyond the forests and the trees to consider local institutions, that is, the rules and strategies that local people develop to organize their relationship with their forest environments. Our analysis suggests that there is a clear and positive correlation between strong local institutions, collective activities and good forest conditions. Government officials need to appreciate the role of local institutions in forest governance when they transfer forest management and ownership to local communities, and they must ensure that appropriate support is given to institutional development in communities that assume these responsibilities.

Social scientists have developed useful tools for analyzing the factors that influence the effectiveness of local user groups in forest management. However, research on communal forestry management should be conservative when generalizing results, since the combination of resources and user attributes varies from situation to situation. Often, simply copying policy solutions from one biophysical, socioeconomic and cultural setting to another does not work. Careful analysis is needed to understand what makes some institutional systems more effective than others.

Research carried out under the IFRI Research Programme provides a useful approach to studying community forestry institutions, as it draws on broad, reliable and comparable sets of time-series data from a large number of community-managed forests around the world. Another of the strengths of the programme is that it provides a systematic way of studying how people interact with forest resources, permitting the measurement of the impact that communities have on forests. Although in this study we used foresters' opinions to gauge forest conditions, we are currently working with a group of forest ecologists to develop methods that will allow IFRI scholars to compare forest conditions based on more objective and measurable forest data. We believe that collaboration between social and natural scientists is key to gaining a better understanding of the role of local institutions in efforts to improve forest governance. Interdisciplinary researchers concerned with human–ecological interactions have a great deal of challenging and exciting work ahead as they study the world's forests and their users.

References

- Agrawal, A. 2001. Common property institutions and sustainable governance of resources. *World Dev.* 29:1649-72.
- Agrawal, A. and J. Ribot. 2000. Analyzing decentralization: a framework with south Asia and east African environmental cases. World Resources Institute, Washington, DC, USA.
- Agrawal, A. and G. Yadama. 1997. How do local institutions mediate market and population pressures on resources? Forest Panchayats in Kumaon, India. *Dev. Change* 28(3): 435-465.
- Andersson, K.P. 2002. Can decentralization save Bolivia's forests? An institutional analysis of municipal forest governance. School of Public and Environmental Affairs and Department of Political Science, Indiana University. Bloomington, USA.
- Andersson, K.P. 2004. Who talks with whom? The role of repeated interactions in decentralized forest governance. *World Dev.* 32(2):233-250.
- Arnold, J.E.M. 1990. Social forestry and communal management in India. Rural Development Forestry Network (RDFN), Overseas Development Institute (ODI), London, UK.
- Arnold, J.E.M. 1992. Community forestry: ten years in review. Community Forestry Concept Note. Food and Agriculture Organization of the United Nations (FAO), Rome, Italy.
- Arnold, J.E.M. 1998. Managing forests as common property. Food and Agriculture Organization of the United Nations (FAO) Forestry Paper. FAO, Rome, Italy.
- Ascher, W. 1995. Communities and Sustainable Forestry in Developing Countries. ICS Press, San Francisco, USA.

- Baland, J.-M. and J.-P. Platteau. 1996. Halting Degradation of Natural Resources: Is There a Role for Rural Communities? Clarendon Press, Oxford, UK.
- Becker, D. and R. Leon. 2000. Indigenous forest management in the Bolivian Amazon: lessons from the Yuracaré people. *In* People and Forests: Communities, Institutions and Governance, (C. Gibson, E. Ostrom and M.A. McKean, eds.). MIT Press, Cambridge, USA.
- Berkes, F. (ed.). 1989. Common Property Resources: Ecology and Community-Based Sustainable Development. Belhaven Press, London, UK.
- Bhatt, C. 1990. The Chipko Andolan: forest conservation based on people's power. *Environ. Urban.* 2:7-18.
- Bromley, D.W. 1991. Environment and Economy: Property Rights and Public Policy. Blackwell Press, Oxford, UK.
- Bromley, D.W., D. Feeny, M. McKean, P. Peters, J. Gilles, R. Oakerson, C.F. Runge and J. Thomson, (eds.). 1992. Making the Commons Work: Theory, Practice, and Policy. ICS Press, San Francisco, USA.
- Clugston, R. and T. Rogers. 1995. Sustainable livelihoods in North America. *Development* 3 (September):60-3.
- Contreras, H.A. and T.M. Vargas. 2001. Dimensiones sociales, ambientales y económicas de las reformas en la política forestal de Bolivia. Proyecto de Manejo Forestal Sostenible and the Center for International Forestry Research, Santa Cruz, Bolivia.
- Dei, G. 1992. A forest beyond the trees: tree cutting in rural Ghana. *Hum. Ecol.* 20(1):57-88.
- Douglass, M. 1992. The political economy of urban poverty and environmental management in Asia: access, empowerment and community-based alternatives. *Environ. Urban.* 4(2):9-32.
- Feeny, D., F. Berkes and J.M. Acheson. 1990. The tragedy of the commons: 22 years later. *Hum. Ecol.* 18:1-19
- Feeny, D., F. Berkes, B.J. McCay and J.M. Acheson. 1998. The tragedy of the commons: 22 years later. *In* Managing the Commons, (J.A. Baden and D.S. Noonan, eds.). Indiana University Press, Bloomington, USA.
- Ghai, D. 1993. Conservation, livelihood and democracy: social dynamics of environmental change in Africa. *Osterreichische Zeitschrift für Soziologie* 18:56-75.
- Gibson, C. and F. Lehoucq. 2003. The local politics of decentralized environmental policy in Guatemala. *J. Environ. Dev.* 12(1):28-49.
- Gibson, C.C., E. Ostrom and M.A. McKean. 2000. Forests, people, and governance: some initial theoretical lessons. Pp. 227-242, *in* People and Forests: Communities, Institutions and Governance, (C. Gibson, E. Ostrom, and M.A. McKean, eds.). MIT Press, Cambridge, USA.
- Gibson, C.C., J. Williams and E. Ostrom. 2005. Local Enforcement and Better Forests. *World Dev.* 33(2):273-284.
- Hernández, I. and D. Pacheco. 2001. La ley INRA en el espejo de la historia: Dos siglos de reforma agraria en Bolivia. Fundación Tierra, La Paz, Bolivia.
- IFRI (International Forestry Resources and Institutions) Research Programme. 2002. Field Manual. Center for the Study of Institutions, Populations, and Environmental Change, Indiana University, Bloomington, USA.
- McCay, B.J. and J.M. Acheson (eds.). 1987. The Question of the Commons: The Culture and Ecology of Communal Resources. University of Arizona Press, Tucson, USA.
- McCay, B. and J.M. Acheson. 1990. Human ecology of the commons. *In* The Question of the Commons: The Culture and Ecology of Communal Resources, (B.J. McCay, and J.M. Acheson, eds.). The University of Arizona Press, Tucson, USA.
- McKean, M.A. 1992. Success on the commons: A comparative examination of institutions for common property resource management. *J. Theor. Polit.* 4(3):247-82.
- McKean, M.A. 2000. Common property: what is it, what is it good for, and what makes it work? Pp. 27-56, *in* People and Forests: Communities, Institutions and Governance (C.C. Gibson, M.A. McKean and E. Ostrom eds.). MIT Press, Cambridge, USA.
- Ostrom, E. 1990. Governing the Commons: The Evolution of Institutions for Collective Action. Pp. 298. Cambridge University Press, New York, USA.

- Ostrom, E. 1992a. Crafting institutions for self-governing irrigation systems. ICS Press, San Francisco, USA.
- Ostrom, E. 1992b. The rudiments of a theory of the origins, survival, and performance of common-property institutions. Pp. 293-318 *in* Making the Commons Work: Theory, Practice and Policy (D.W. Bromley ed.). ICS Press, San Francisco, USA.
- Ostrom, E. 1998. The international forestry resources and institutions research programme: a methodology for relating human incentives and actions on forest cover and biodiversity. Pp. 1-28 *in* Forest Biodiversity in North, Central and South America, and the Caribbean: Research and Monitoring. Man and the Biosphere Series, Vol. 21 (F. Dallmeier and J.A. Comiskey, eds.). United Nations Educational, Scientific and Cultural Organization (UNESCO), Paris, France; Parthenon, New York, USA.
- Ostrom, E. 1999. Self-governance and forest resources. CIFOR Occasional Paper 20 (February):1-15. Centre for International Forestry Research (CIFOR), Bogor, Indonesia.
- Ostrom, E. 2003. How types of goods and property rights jointly affect collective action. *J. Theor. Polit.* 15(3):139-170.
- Ostrom, E. and V. Ostrom. 1999. Public goods and public choices. Pp. 75-105 *in* Polycentricity and Local Public Economies (M. McGinnis ed.). The University of Michigan Press, Ann Arbor, USA.
- Ostrom, E. and J. Walker. 1997. Neither markets nor states: linking transformation processes in collective action arenas. Pp. 35-72 *in* Perspectives on Public Choice: A Handbook (D.C. Mueller ed.). Cambridge University Press, Cambridge, UK.
- Ostrom, E., L. Schroeder and S. Wynne (eds.). 1993. Institutional Incentives and Sustainable Development. Westview Press, Boulder, USA.
- Ostrom, E., C. Gibson, S. Shivakumar and K. Andersson. 2002. Aid, incentives and sustainability: Institutional analysis of international development cooperation. SIDA Studies in Evaluation No. 02/01. Swedish International Development Cooperation Agency (SIDA), Stockholm, Sweden.
- Pacheco, P. 1999. Estilos de desarrollo, deforestación y degradación de los bosques en las tierras bajas de Bolivia. CIFOR (Centre for International Forestry Research), CEDLA (Centre for Latin American Research and Documentation), Fundación Tierra, La Paz, Bolivia.
- Perry, J. and R. Dixon. 1986. An interdisciplinary approach to community resource management: Preliminary field test in Thailand. *J. Develop. Area* 21(1):31-47.
- Peters, P. 1994. Dividing the Commons: Politics, Policy and Culture in Botswana. University of Virginia Press, Charlottesville, USA.
- Poteete, A. and E. Ostrom. 2002. In pursuit of comparable concepts and data about collective action. Presented at the CAPRI Workshop on Methods for Studying Collective Action, Nyeri, Kenya. Available from the Workshop in Political Theory and Policy Analysis, Indiana University. Bloomington, USA.
- Raju, G., R. Vaghela and M. Raju. 1993. Development of People's Institutions for Management of Forests. Viksat, Nehru Foundation for Development, Ahmedabad, India.
- Richards, J.F. and R.P. Tucker. 1988. World Deforestation in the Twentieth Century. Duke University Press, Durham, USA.
- Scott, J. 1998. Seeing like a State. How Certain Schemes To Improve The Human Condition Have Failed. Yale University Press. New Haven, USA.
- Swaney, J.A. 1990. Common property, reciprocity and community. *J. Econ. Issues* 24(2):451-462.
- Tucker, C., E. Salvador and K. Andersson. 1999. Resumen preliminar del trabajo de campo de CIPEC en Capulalpam, Oaxaca, Mexico, 3-27 Mayo 1999. Centre for the study of Institutions, Population and Environmental Change (CIPEC).
- Urioste, M. and D. Pacheco. 2001. Las Tierras Bajas de Bolivia a Fines del Siglo XX: Tenencia, Acceso y Uso a la Tierra y los Bosques. PIEB (Programa de Investigación Estratégica en Bolivia), La Paz, Bolivia.

- Varughese, G. 1999. Villagers, Bureaucrats and Forests in Nepal: Designing Governance for a Complex Resource. Political Science and SPEA, Indiana University, Bloomington, USA.
- Wade, R. 1994. Village Republics: Economic Conditions for Collective Action. Cambridge University Press, Cambridge, USA.
- Wiesner, B. 1990. Harvest of sustainability: recent books on environmental management. J. Dev. Stud. 26:335-41.
- Winter, M. 1998. Decentralized natural resource management in the Sahel. Associates of Rural Development. Burlington, USA.
- Wunsch, J.S. and D. Olowu (eds.). 1995. The Failure of the Centralized State: Institutions and Self-Governance in Africa (2nd edn.). ICS Press, San Francisco, USA.

Chapter 3

Extraction of nonwood forest products

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1. Introduction

Nonwood forest products (NWFPs) are crucial resources for livelihoods in many parts of the world. This chapter addresses issues related to the potential of NWFPs for income generation at local, national and global levels. It looks at how the use of and trade in NWFPs affects the sustainability of different extraction regimes and livelihood strategies, with a focus on South America and especially Brazil. Challenges to define NWFPs and to monitor sustainability of their use are highlighted with recommendations for further research.

2. Terminology

Many terms are used to capture the wide range of forest-based plants and animals from which goods (other than timber or wood) and services are derived. The oldest terms are 'minor forest products', 'secondary forest products', 'naval stores' and 'forest by-products'. By the 1970s, these terms were supplemented with 'nonwood forest products' and 'nontimber forest products'. Since the 1990s, new terms have emerged including 'special forest products', 'wild crafted products', 'biodiversity products', 'natural products', 'nonwood goods and benefits' and 'tree-crops'.

Why have these terms emerged? Deforestation in the tropics accelerated in the early 1980s and timber exploitation practices began to be perceived as overly destructive. Gradually, more emphasis was given to the interests of forest-dependent people in developing countries and to the importance of forest products other than timber. The sustainable use of a broader range of forest plant and animal species was receiving more attention, particularly from nature conservation agencies as a way of mitigating deforestation, while at the same time increasing people's income. To some degree, the terms were coined in order to facilitate a shift in emphasis towards the livelihoods of forest-dependent peoples and to more 'environmentally friendly' uses of forests, with the objective of encouraging more balanced uses of forest resources and a reduction in forest degradation and deforestation (see Box 1, overleaf, for an extended explanation).

While the emphasis on promoting NWFPs was at first placed on tropical forests and on developing countries, awareness is now growing of their similar importance in temperate and boreal forests of the developed countries.

The acronym and term NWFP will be used throughout this text for reasons of consistency and clarity.

3. Socioeconomic importance of nonwood forest products

3.1 Household level

NWFPs are of primary significance for subsistence and/or income at the household and village levels in rural forested areas. Forest-dependent people are generally those who rely most on NWFPs, but many more people are also seasonally dependent on them.

Box 1. 'Extrativismo', a Portuguese term for NWFP

Since colonial times, the Portuguese have used the term 'extrativismo' to group all those products that were obtained from gathering and hunting activities in their vast forest reserves in Brazil. These products include animal furs, fruits, nuts, latexes, medicines, timber, fuelwood, charcoal and natural dyes like the red dye extracted from the wood of *Caesalpinia echinata*. This dye was removed by heating the wood over a glowing brazier of charcoal, called a 'brasa' in Portuguese, which gave its name to the tree 'pau Brazil' and, subsequently, to the country.

Today, the term 'extrativismo' is still used in the Amazon and by the national Brazilian Statistical Agency (IBGE) to classify the 'produção da extração vegetal', that is, the quantities and values of wood and nonwood products obtained from the exploitation of native forests (IBGE 2003, see web link: <http://www1.ibge.gov.br>). For wood and nonwood products obtained from planted forests, mostly using non-indigenous or exotic species, IBGE employs the term 'produção da silvicultura' that includes NWFPs like pine resins, eucalyptus leaves or bark from black wattle (*Acacia mearnsii*). Thus, by linguistically separating products gathered in natural forests from those from plantations, the IBGE statistics highlight the origin of their production.

Seasonal examples include livestock browsing in forests during the dry season and displaced people during times of famine, floods or war. NWFPs thus contribute to the diverse survival options of rural people by providing buffers against environmental and economic adversities. NWFPs are also important elements of cultural, religious, spiritual and recreational needs. As NWFP use is mainly informal, there is a huge lack of reliable data on their production and trade and on the number of people involved, which makes it hard to assess the effective contribution of NWFPs to rural livelihoods.

Moreover, most of the literature dealing with NWFPs describes and analyzes specific products or issues within small areas used by well-defined groups of people over short periods of time. One example of such a study is provided by Zitzmann (1999), who undertook a comprehensive analysis of NWFP impact on local economic strategies in Botswana. In comparing NWFP extraction with other income-generating activities, it was found that income derived from selling *Imbrasia belina* caterpillars represented 13% of the household total cash income per year, with the associated labour input representing only 5.7% of all income-generating activities. The sale of *Imbrasia belina* was therefore of great importance to local livelihoods. Nevertheless, although interesting, these results describe a rather small case study that is hard to generalize. Further problems in generalizing case studies like this stem from difficulties in understanding how NWFP-based economic strategies have evolved over time, particularly in rural areas that are experiencing rapid economic change owing to increased road access, rural population exodus and land clearing for farming, forest logging or mining.

Nevertheless, some recent NWFP studies have tried to develop larger comparative frameworks. The ongoing study carried out by the Centre for International Forestry Research – CIFOR (Belcher and Ruiz-Peres 2001; WFC NWFP side event 2003 [see Box 6]; Ruiz-Pérez *et al.* 2004) compares the importance of different NWFPs to local livelihoods over a larger sample of case studies from developing tropical countries. The study is creating an analytical framework in order to facilitate comparative analyses of different extraction regimes for different products within a range of developing countries. The framework is designed to help identify those parameters that are critical to our understanding of how NWFPs contribute to people's livelihoods over time and space.

Thus, when analyzing NWFP-based livelihoods at the local level, it will always be preferable to review them over larger populations/areas and over longer time frames. This will be essential to assessing their long-term effects, related sustainability issues

and market trends, and especially when comparing NWFP gathering with other income-generating activities in rural areas such as farming, fishing, tourism, mining or industry.

3.2 Country level

Although the importance of NWFPs at the local level is becoming better documented, synthetic information at country levels is rather scarce. While dispersed information from local-level use of NWFPs now forms the basis of most associated policy creation and decision-making in rural development programmes of governments and international agencies, NWFP information remains inadequately incorporated into macro-economic indicators.

For example, while some species furnish substantial amounts of raw material for industrial processing at national levels, such as pine resins in China, cork in Portugal, Arabic gum in Sudan, rattan in the Philippines, bamboo in Indonesia, and medicinal plants in India, only a few countries systematically monitor production and trade of their most important NWFPs. Reliable data on NWFP sources and the amount of labour involved in extraction and transformation are also often lacking. When these data do exist, they are often mixed with agricultural production information and/or are limited to internationally traded products.

Many reasons can be advanced to explain why reliable data on NWFPs are difficult to obtain. Among these is the fact that many (sometimes scores of) products are involved, with these usually being gathered by large numbers of small-scale producers and scattered over large (and often remote) areas. In addition, the relatively low values of NWFP production when compared to those from agricultural crops or wood products is a serious disincentive to creating and maintaining quality NWFP data-acquisition programmes. Technical difficulties and the high cost of surveying and assessing the acquisition and transformation of NWFPs make it difficult for most countries to obtain reliable data on them.

All these factors combine to make it difficult to assess the reliability of NWFP data when they exist, to monitor trends in production over time, and to compare statistics across countries. Even determining the economic value of NWFPs can be difficult when prices are reported at different stages of production and processing.

An example of the national production levels and values of major NWFPs can be found in the Annual Statistical Yearbooks of Brazil, produced by IBGE (Tables 1 and 2, on pp. 54 and 55).

In countries where reasonably reliable statistics are available, like these from Brazil, the economic contribution of NWFPs is usually found to be much lower than that derived from wood production. The total production value of Brazilian NWFPs for 2001 amounted to R\$445 451 000, while wood-related forest production amounted to R\$3 726 358 000. Using an average exchange rate in 2001 of US\$1.00 = R\$2.30, the value of Brazilian NWFPs was approximately US\$193 million and the value of log production was approximately US\$1620 million. In comparing the reported values of wood and NWFP outputs, we see then that NWFPs represent about 10.6% of total forestry production with logs accounting for about 89.3%.

Looking at trends over time, Brazil provides a remarkable example of a significant decline in NWFP production over the last 30 years (Table 3, on p. 55), despite the efforts of governmental and nongovernmental nature conservation agencies to promote NWFP extraction and commercialization. Over the same period, wood production in Brazil increased substantially.

The causes suggested for this decline in Brazilian NWFP production include: (i) deforestation; (ii) migration of rural people to urban areas, resulting in fewer people collecting NWFPs; (iii) low prices for NWFPs; and (iv) reduced markets for some products as alternative cheaper sources or surrogates became available (e.g., the supply of natural latex now largely comes from industrial plantations of *Hevea brasiliensis* outside the Amazon region).

Nevertheless, Brazilian NWFP production trends reported by IBGE over the last three decades should not be used as an example of a worldwide trend, but should serve as a

Table 1. Plant products harvested from Brazilian forests in 2001.

Plant products	Quantity (tonnes)	Value (Real\$ x 1000)
Rubbers		
<i>Hevea</i> (latex coagulated)	4 368	6 819
<i>Hevea</i> (latex liquid)	131	93
Maçaranduba	4	15
Sorva	39	61
Waxes		
Carnaúba (wax)	2 883	9 776
Carnaúba (powder)	12 315	15 953
Fibres		
Buriti	356	192
Carnaúba	1 384	465
Piaçava	95 447	109 802
Others	71	30
Tannins		
Angico (bark)	347	93
Barbatimão (bark)	12	5
Others	5	4
Oils		
Babaçu (kernels)	114 563	43 595
Copaíba (oil)	414	1 056
Cumaru (kernels)	38	64
Licuri (shells)	5 106	2 203
Oiticica (seeds)	249	27
Pequi (kernels)	3 338	2 971
Tucum (kernels)	796	239
Others	392	218
Foods		
Açaí (fruit)	123 135	66 845
Cashew nuts	6 266	4 128
Brazil nuts	28 467	27 695
Erva-Maté Cancheada	182 177	109 179
Mangaba (fruit)	1 181	520
Palmito	15 596	9 093
Pine seeds	4 417	2 996
Umbu (fruit)	9 919	3 498
Aromatic and medicinal plants		
Ipecacuanha (roots)	1	2
Jaborandi (leaves)	1 146	1 319
Urucu (seeds)	178	251
Others	5 446	3 248
Others		
Black Wattle dry bark	212 425	7 303
Eucalyptus leaves	54 043	3 349
Pine resins	28 366	12 344
Total nonwood value		Real\$445 451

Source: IBGE 2003.

Table 2. Wood products harvested from Brazilian forests in 2001 (IBGE 2003, see web link: <http://www1.ibge.gov.br>).

Wood production from natural forests	Quantity	Value (Real\$ x 1000)
Charcoal (1)	1 729 319	226 646
Fuelwood (2)	49 001 583	324 008
Logs (2)	20 069 287	995 283
Knots (3) (<i>Araucaria angustifolia</i>)	305 479	2 400
Wood production from plantations		
Charcoal (1)	2 092 309	338 996
Fuelwood (2)	30 042 485	328 888
Logs	69 758 138	1 510 137
for pulp (2)	40 999 323	730 243
other (2)	28 758 815	779 894
Total wood value		3 726 358

Table adapted by the author and reorganized into categories of 'wood production from natural forests' and 'wood production from plantations'. (1) Quantities estimated in tonnes, (2) quantities estimated in cubic metres, (3) quantity in units. Values are prices paid to producers.

Table 3. Production of major NWFPs in Brazil during selected years from 1970–2001.

Product	Quantity (tonnes)		Quantity (tonnes)		Value (Real\$ x 1000)	
	1970	1980	1990	1998	2001	2001
<i>Hevea</i> (latex coagulated)	51 984	21 250	22 896	5 449	4 368	6 819
<i>Hevea</i> (latex liquid)	6 324 *	2 571	1 109	659	131	93
Maçaranduba	595	406	116	4	4	15
Sorva	4 692	3 220	736	37	39	61
Piaçava fibres	21 654	55 939	59 381	96 413	95 447	109 802
Babaçu nuts	180 897	250 951	188 718	122 077	114 563	43 595
Açaí fruits	nl	59 591	120 795	119 074	123 135	66 845
Brazil nuts	104 487	40 458	51 195	23 111	28 467	27 695
Maté leaves	113 460	105 004	150 823	183 504	182 177	109 179
Ipecacuanha roots	53	14	2	1	1	2
Palmito	24 625 **	114 408	27 031	24 188	15 596	9 093
Copaiba oil	nl	nl	nl	398	414	1 056
Pine resins	nl	nl	nl	nl	28 366	12 344
Total number of NWFPs listed	30	52	32	29	29	

Compiled by author from IBGE yearbooks (IBGE 1972, 1982, 2000, 2003, see web link at <http://www1.ibge.gov.br>). 'Value' refers to prices paid to producers; nl = not listed; * = 1971; ** = 1978.

Box 2. The most important NWFPs from South America

Apart from fuelwood and charcoal, the most important NWFPs in South America are foods and drinks such as Brazil nuts, fruits, palm hearts and palm wines, mushrooms and maté, resins, latexes and essential oils, medicinal plants, fibres, construction materials, fodder, colorants and tannins.

This box illustrates a few examples of different NWFP uses.

In Amazonia, the best-known edible products are Brazil nuts and palm hearts. Both can be found in domestic, regional and international markets. Brazil nuts are still collected almost exclusively from wild sources of *Bertholletia excelsa* in Bolivia, Brazil and Peru.

Palm heart production is more widespread, occurring mainly in tropical areas of Brazil, Bolivia, Colombia, Venezuela, Guyana and Peru. Palm hearts are extracted from wild stands of *Euterpe oleracea* and *E. precatoria* or from cultivated species like *Bactris gasipaes*. The fruits of these palm species also play important roles in food and drink in the Amazon region. For edible seed and industrial oil production, other important palm species for both subsistence and commerce are *Orbignya phalerata*, *Mauritia flexuosa* and *Jessenia bataua*.

The tree species *Platonia insignis*, *Myrciaria dubia*, *Theobroma grandiflorum* and *Couepia longipendula* also produce edible fruits or nuts of local importance. The seeds of *Araucaria angustifolia* in Argentina and southern Brazil and of *Araucaria araucana* in Argentina and Chile are commonly consumed by people and fed to livestock.

In Argentina, Uruguay, Paraguay and southern Brazil, the leaves of *Ilex paraguariensis* are used to brew maté, an extremely popular tea-like beverage. Though wild in native forests of the Alto Paraná region, the Alto Uruguay region and northeastern Argentina, it is now cultivated on a large scale in plantations, especially in Argentina and Brazil.

Latex extracted from the native Amazonian *Hevea brasiliensis* is used for the production of natural rubber. Other important exudates from tropical South America are jatobá (*Hymenaea courbaril*), maçaranduba (*Manilkara huberi*), sorva (*Couma* spp.), balata (*Manilkara bidentata*) and balsamo (*Myroxylon balsamum*). Copaiba (*Copaifera* spp.) and dragon's blood (*Croton dracooides*) are used in local medicine.

South America has a long tradition of medicinal plant use. One of the great legacies of the South American people is the bark derived from *Cinchona* species, the source of the antimalarial drug quinine. World production of quinine bark is approximately 8000–10 000 tonnes per year. Brazil, Bolivia and Colombia are important producers of quinine in South America.

Peru is the world's largest producer of fruits from the tara tree (*Caesalpinia spinosa*) for the extraction of tannins. Production is mainly from natural stands, but some trees are harvested from agroforestry systems. *Caesalpinia* forests are most extensive in Peru, followed by Bolivia and to a lesser extent Chile, Ecuador and Colombia. *Quebracho colorado* (*Schinopsis* spp.) is another source of tannin in Argentina and Paraguay.

Examples of other uses of NWFPs include the extraction of fibres from the following species: the palms *Carludovica palmata* in Panama and Ecuador, *Attalea funifera* and *Leopoldina piassaba* in Brazil. *L. piassaba* is harvested on a smaller scale in Venezuela and Colombia for uses ranging from hats to brooms.

The bamboos *Guadua angustifolia* and *Chusquea* spp. are widely employed in construction, furniture and handicrafts in Ecuador, Colombia and Venezuela.

Source: Forest Resources Assessment 2000 (FAO 2001a).

warning that expectations should not be raised too high with regard to significant and continuous contributions of NWFPs to the national income of many countries. This partly explains the low interest levels of policy-makers in several countries in creating incentives for developing the NWFP sector or to support programmes for NWFP data collection. Box 2 (opposite) has more information on South American NWFPs.

NWFPs have recently attracted the attention of international development agencies and conservation-oriented nongovernmental organizations (NGOs) because of the environmental services and social benefits that they may provide. This has raised expectations among a wide range of nature conservation and forest management agencies that NWFPs and their commercialization could lead to forms of 'benign' forest utilization, possibly even creating incentives for forest conservation. Conservation organizations have been prominent among these advocates for the exploitation of NWFPs because they see them as a potential way of supporting local and regional sustainable development while simultaneously promoting forest conservation.

3.3 International level

Some NWFPs are export commodities and are significant in international trade. Most of these products are exported in raw or semiprocessed states. Examples are rattan, bamboo, cork, forest nuts (see Box 3, on p. 59) and mushrooms, gum Arabic, essential oils and medicinal plants (Table 4, overleaf).

Table 4 lists 28 commodities covering a range of major NWFPs traded at the international level and for which specific HS codes exist. The Harmonized Commodity Description and Coding System, generally referred to as the 'Harmonized System' or simply 'HS', is a multipurpose international product nomenclature developed by the World Customs Organization (WCO), headquartered in Brussels, Belgium. It comprises about 5 000 commodity groups, each identified by a six digit code, and arranged in a legal and logical structure that is supported by well-defined rules to achieve uniform classification. The system is used by more than 190 countries as a basis for their customs tariffs and for the collection of international trade statistics. More than 98% of the merchandise in international trade is classified in terms of the HS (<http://www.wcoomd.org>). The trends in trade for the Table 4 NWFPs between 1992 and 2002 suggest only a modest increase in total absolute trade value, while total value of world trade increased 2.5 times over the same period. It is interesting to note that many of the products with substantial increases in trade value between 1992 and 2002, such as natural cork, mosses and lichens for bouquets, truffles, mushrooms, chestnuts, bamboo, palm hearts and maple syrup, originate primarily in developed countries.

The impact of international trade in NWFPs on poverty alleviation needs further study. Once trade goes beyond local and regional markets, it becomes more sophisticated in the sense that it requires capital investment and a wider range of skills that are usually not available to rural producers in developing countries. If the product is to be traded internationally, a number of import-export operations must occur and be paid for. These include fulfilling export and import requirements such as respecting quality standards and phytosanitary regulations, payment of permits and taxes, and storage, processing and transport involving an array of agents and distributors. All this must occur before a more highly processed and sometimes completely transformed product can be sold by retailers to consumers. Appreciation of and access to such systems can be intimidating if not impossible for producers of NWFPs. Indeed, access to international markets will almost always require a substantial level of organization from producers in order to ensure that they do not become marginalized in the process. This is particularly true for products from developing countries, and this seems to be reflected by the stagnating or declining trade values of many NWFPs as seen in Table 4.

Nevertheless, at the international level, consumers are increasingly concerned that trade is 'fair' to all involved. This can mean promoting conditions or practices that safeguard the interests of less-powerful partners, typically producers, in trade chains, or working to increase their ability to exert power by promoting producer associations and networks. 'Fair trade', a common term applied to those schemes favouring more social

Table 4. Global import values of key NWFPs for 1992 and 2002 (in US\$ x 1000).

HS Code	Commodity description	Global import value	
		1992	2002
060410	Mosses and lichens for bouquets and ornamental purposes	9 352	25 476
070952	Truffles, fresh or chilled	4 201	23 656
070959	Mushrooms other than <i>Agaricus</i> , fresh or chilled	n.a.	364 412
071239	Mushrooms (excl. 071331/33) and truffles, dried	n.a.	219458
200320	Truffles, prepared or preserved, not in vinegar	3 049	11 012
080120	Brazil nuts, fresh or dried	44 344	59 848
080240	Chestnuts, fresh or dried	109 958	184 663
230810	Acorns and horse-chestnuts for animal feed	1 216	7 380 ¹
120792	Shea nuts (karite nuts)	5 155	5 136 ¹
121110	Liquorice roots	33 455	24 310
121120	Ginseng roots	389 345	221 435
121190	Plants and parts, for use in pharmaceuticals, perfumes and insecticides	689 926	777 980
121210	Locust beans, locust seeds	22 395	40 239
130110	Lac	25 286	25 653
130120	Gum Arabic	101 312	105 510
130190	Natural gum, resin, gum-resin, balsam, not gum Arabic	92 755	96 535
400130	Balata, gutta-percha, guayule, chicle and similar gums	26 726	13 605
130214	Pyrethrum, roots containing rotenone, extracts	27 865	26 173 ¹
140110	Bamboos used primarily for plaiting	37 562	50 054
140120	Rattan used primarily for plaiting	118 987	51 327
140210	Kapok	11 920	2 826 ¹
170220	Maple sugar and maple syrup	43 632	116 202
200891	Palm hearts, otherwise prepared or preserved	16 082	67 514
320110	Quebracho tanning extract	51 938	45 173
320120	Wattle tanning extract	63 877	34 168
320130	Oak or chestnut extract	8 653	917 ¹
450110	Natural cork, raw or simply prepared	7 874	110 702
530521	Abaca fibre, raw (<i>Musa textilis</i>)	15 221	20 374

Compiled by author based on Comtrade data (<http://unstats.un.org/unsd/comtrade>)

n.a. = not applicable, this code did not exist in the HS 1992 version.

¹ 2001

Box 3. International trade in Brazil nuts

Bolivia and Brazil dominate world production and export of Brazil nuts (Table 5). The United States of America is the world's largest importer with a market share of 33%, followed by the United Kingdom (19%), Germany (9%) and the Netherlands (6%). Demand and price movements for Brazil nuts tend to be seasonal, with unshelled demand particularly high during the Christmas period. However, demand can vary because Brazil nuts are easily replaced by comparable nuts such as almonds, cashews, macadamias or hazelnuts. The principal forces influencing harvest levels in Amazonia are price and demand fluctuations of comparable nuts in international markets. As a result, annual production of Brazil nuts fluctuates substantially. For example, Brazil produced a record level of 104 487 tonnes in 1970 while in 1998 it produced only 23 111 tonnes (Table 3). During the same period, cashews, almonds, macadamias, pecans, hazelnuts and walnuts saw their production quantities and trade values significantly increase.

The Brazil nut tree (*Bertholletia excelsa*) or 'Castanheira' in Portuguese is one of the largest trees of the Amazon forest. While also providing good timber, the tree has nuts that are gathered in forests by local people who sell them to traders for further processing in urban centres. Nearly all Brazil nut production comes from wild sources, and the majority of this is for export. Total annual world exports fluctuate between 30 000 and 50 000 tonnes, with values ranging from US\$39.6 million up to 78.5 million (Table 6).

Table 5. Major exporters of Brazil nuts, HS 080120, in 2001.

	Export in 2001			
	Value	%	Quantity	%
Bolivia	26 561	52	13 343	38
Brazil	11 150	22	10 552	30
Peru	3 433	7	1 910	5
Indonesia	2 860	6	4 871	14
Netherlands	1 967	4	748	2
Italy	1 180	2	104	0.3
Rest of World	3 590	7	3 621	10.7
World	50 741	100	35149	100

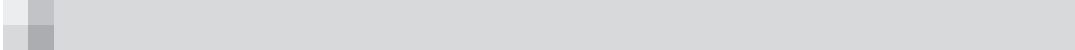
Source: Comtrade 2003 (<http://unstats.un.org/unsd/comtrade>). Value in US\$ x 1000, quantity in tonnes

Table 6. Total world export of Brazil nuts, HS code 080120.

Year	Total world export	
	Value	Quantity
1995	39 695	39 474
1996	58 541	31 964
1997	75 462	46 931
1998	65 130	44 423
1999	65 273	28 787
2000	78 514	50 695
2001	50 741	35 149

Value in US\$ x 1000; quantity in tonnes.

Source: Comtrade 2003 (<http://unstats.un.org/unsd/comtrade>)



equity in trade (<http://www.fairtradefederation.com/>), and other types of trade promotion instruments, such as forest certification, offer opportunities for NWFP producers to enter and benefit from international trade (Box 4, opposite).

While the certification of agricultural and wood products is well advanced with several schemes firmly in place (Box 5, on p. 62), the development of technical criteria and standards for certification of NWFPs is still in its infancy. Furthermore, certification schemes are usually set up to regulate international markets, while NWFPs are still traded mostly at local and regional levels. Although the market share of organic, fair-traded, or otherwise certified products is increasing, the contribution of NWFPs is still very limited. For instance, in consulting trade data on organic agricultural products in 34 countries worldwide gathered by Stiftung Ökologie und Landbau, Germany (2003), no reference to NWFPs was found (http://www.soel.de/publikationen/oel/oel_inhalt2003.html).

Increased globalization of trade favours the cheapest suppliers, whatever their location, and this can have a devastating effect on the emerging activities of small-scale NWFP producers. Furthermore, international trade usually requires a large continuous product supply, which may not be achievable by NWFP producers unless they can somehow expand their production beyond natural forests. Moreover, many species that are of economic interest to producers of NWFPs have limited geographic distributions, often occurring in one or a few countries only, while their markets are global.

4. Assuring a sustainable supply of nonwood forest products for trade

The foremost issue in the successful and sustainable development of NWFPs for commercial use is resource availability. Subsistence use of NWFPs will in most cases not result in serious resource management challenges nor in resource depletion, apart from exceptional cases that can occur during droughts, floods or war. It is the commercial exploitation of NWFPs that leads to supply constraints, and this usually occurs in the context of free access systems where the resource is harvested in uncontrolled and unlimited quantities.

Forest species that yield NWFPs usually occur in low frequencies, especially in tropical forests, and these species are unlikely to become important suppliers of commercially large quantities, as they can be quickly overharvested. On the other hand, some NWFP species are common, like açai palms (*Euterpe* spp.) in the Varzea forests of the lower Amazon River, or the pines that occur in South American coniferous forests and produce resins. Mushrooms and caterpillars can sometimes be very common under favourable climatic conditions. Resource constraints are not a major limiting factor for trade in cases like these.

Basic information about NWFPs, for example about their biology and population dynamics, or the socioeconomic context of their use, including access and user rights, is often unknown or largely ignored when commercial exploitation of an NWFP is promoted. Actions that address the supply of NWFPs for trade consist mainly of regulating access to the resource, enhancing the resource productivity through forest management, offering economic incentives or a combination of all three.

Market influence on nonwood forest products extraction rates

Whether official or informal, regulating the socioeconomic context of NWFP exploitation is usually the first and easiest step taken when supply constraints appear. Key stakeholders in the market chain, whether producers, traders, consumers or governments, have decisive roles to play. Traders can influence the output of raw materials by increasing prices paid to producers. Through licences or gathering permits, governments can create legal frameworks that regulate access to public forests and monitor harvests. Consumers influence trade by their preferences for products or processes (e.g., organic products or fair trade), while producers can expand or improve their gathering intensity or change their production systems.

Box 4. How trade certification schemes can help NWFP producers

In developing countries, 'fair trade' or forest certification and organic labelling schemes are becoming options to help protect the commercial viability of NWFP-based businesses against competition from similar products obtained through farming or synthetic substitutes. 'Proper' forest management certification schemes, along with schemes that certify 'appropriate' behaviours and procedures in agriculture (e.g., 'proper' labour practices, organic farming, fair trade), offer promising frameworks for successful commercialization of certified NWFPs. Such schemes can help guarantee better prices to gatherers, social equity within the processing and marketing chains, and ensure that attention is given to the environmental sustainability of the resources providing NWFPs. Fair trade markets and organic products offer major advantages to NWFP producers because of the smaller quantities needed to supply international trade (when compared to agriculture-based production), and because premium prices can be offered to producers.

In addition, fair trade food markets and foods labelled organic are among the fastest-growing market sectors in the food distribution sector. Several NWFPs ideally fit such niche approaches, particularly those that have a high per-unit value, a long shelf life, and are easy to process, store and handle. Good examples are essential oils, honey, bamboo, herbs and nuts.

Several types of certification schemes already exist, covering a range of products in agriculture, fishing and forestry, but NWFPs are only marginally involved in these schemes. Some major challenges to increasing certification of NWFPs include:

- **Dispersed nature of production:** Monitoring the NWFP production of many small-scale producers dispersed over large areas is a problem. Difficulties arise in ensuring that products come from certified sites. This can result in higher monitoring costs that may be prohibitive for small-scale producers of NWFPs.
- **Definition of sustainable harvest levels:** Appropriate methodologies and standards to define or verify sustainable harvest levels and practices for many NWFPs are still under development.
- **Resource user conflicts:** Restricting access to certified harvest sites for NWFPs can create conflicts within and among forest-user groups.
- **Unclear market potential:** Actual market demand for certified NWFPs is the driving force behind many certification initiatives and is key to ensuring the economic viability of NWFP production schemes. However, for many NWFPs it is not yet clear that customers will pay premium prices for products certified as fair trade or organic.
- **Insufficient product definition and classification:** Most NWFPs are not yet included in international classification or standardization systems (e.g., the Harmonized System, Standard International Trade Classification, Codex Alimentarius), and this slows their marketability in international trade.
- **Insufficient collaboration/compatibility among existing certification schemes:** The proliferation of certified products is creating confusion among consumers.
- **The limited potential for mainstream benefits of certification schemes to cover all producers in the sector:** For example, the present market share of certified Brazil nuts is only a fraction of the total world production, so the number of Brazil nut gatherers that may benefit from a certification scheme is limited by the market share of the certified product.
- **'Nontariff' trade barriers:** In the areas of food and herbal/medicinal products, trade barriers can generate obstacles for certification efforts. For example, in 2003 the European Union declared a full ban on the import of Brazil nuts from Brazil because of high levels of aflatoxins. Although production from certified places of origin may have acceptable levels of aflatoxins, the certified products are equally affected by the ban.

Source: Vantomme and Walter 2003.

Box 5. Types of certification schemes applicable to NWFPs

Traded NWFPs can be certified according to different objectives; the most important are:

1. **Improvement in social equity.** This occurs through social certification such as fair and ethical trade. Fair Trade Labelling Organizations International is an example;
2. **Environmental concerns.** These are addressed through forest-related certification schemes guaranteeing that the products are obtained from sustainably managed forests. The Forest Stewardship Council (FSC) and the Pan European Forest Certification (PEFC) are examples;
3. **Product quality certification,** particularly for food products, and for which trade is governed by national, regional and international food control agencies. At the international level, trade in food products occurs according to international standards and specifications provided by agencies like the International Organization for Standardization (ISO), Codex Alimentarius (FAO–World Health Organization [WHO]), and Good Manufacturing Practices (GMP). For medicinal plants, Good Harvesting Practices for medicinal plants (WHO) is an example;
4. **Organic certification.** The International Federation of Organic Agriculture (IFOAM) is an example;
5. **‘Certificate of Origin’ schemes.** These are widely used to certify that a product originates from a given region that may have high prestige among consumers. Examples are wine, honey, mushrooms and berries.

The example of fluctuating annual production of Brazil nuts (Box 3) illustrates that the supply of traded NWFPs for this species and for many others depends directly on the prices offered to gatherers. Prices tend to be cyclical, as they depend more on economic, social or climatic factors outside the producers’ region or on the price fluctuations of their competing substitute(s). When higher prices are offered, producers will intensify or expand their gathering efforts over larger areas as appropriate to their means. When prices are down, they may even forsake gathering, as it may not compensate their time investment vis-à-vis other income-earning options. Export bans can also negatively affect prices paid to producers. A good example is the export ban that was placed on raw rattan cane in Indonesia in 1979. This resulted in a price decline for raw cane paid to producers and a fall in raw cane production. The export ban was abolished in 1999 and this liberated prices, and a sharp increase in cane production and exports followed (Purnama *et al.* 1998; Sastry 2001).

Examples like these, whether for Brazil nuts and açai in the Amazon, gum Arabic and shea nuts in the Sahel, or rattan in Indonesia and Malaysia, are similar in that the quantities extracted from forests for trade represent only a fraction of what is still potentially available for harvest. In such cases, price fluctuations will affect the supply much more than forest-use regulations or resource management initiatives, though this is true only in regions still rich in the forest or tree resources that provide NWFPs and when market demands are still relatively modest.

However, the margin for increasing prices to producers is limited, as expanding extraction efforts over larger areas quickly become too expensive and time-consuming for the gatherers. Higher prices for raw NWFP materials make them less competitive in the market than their substitutes. Producers of NWFPs sell their production to intermediate traders who are the key partners in the market chain from producer to consumer, and whose decisions critically affect the NWFP trade. From the trader’s point of view, when their NWFP profit margins decline, they may shift their investments to other products with better margins.

Higher prices for gatherers will also lead to intensified collecting (harvesting more per unit area) and to an expansion in the areas for gathering (harvesting from a larger area).

However, when the accessible forest area is limited, this can lead to stiff competition and conflicts among harvesters and other forest users (e.g., gatherers of wild mushrooms).

Demand for nonwood forest products and ways to regulate extraction

When market demands for NWFPs increase, or the area of accessible forests declines, harvesters, local communities or governments can install informal or formal frameworks that regulate access to their forests and monitor NWFP harvests. This can occur through licences or gathering permits with the objective being to protect both gatherers' income and the resource supply. Governments can play a key role here, as in many countries they are the largest forest owners. Governments also have a social obligation to ensure equitable distribution of benefits among all forest-user groups, but with a particular attention to assist weaker groups of society such as indigenous forest-dependent communities, who usually gather NWFPs.

However, the commercial success of an NWFP at a global scale has the potential to result in such high demand that supply cannot be assured from forest-based production and by forest dependent people alone. Some examples where industrial scale production has developed to meet demand are macadamia nuts in Australia, Hawaii (USA) and South Africa, wild rice in California (USA), and bamboo shoots in China.

High levels of demand for some NWFPs traded internationally will thus require intensification of production to increase the quantity and the quality of the product. A stable and/or growing demand with fair prices offered to producers gives strong incentives to private investors at all levels to increase or improve production through more intense management of the resource, whether this occurs through silviculture, cultivation in farms, or intermediate agroforestry schemes.

From a forester's viewpoint, increasing the resource supply of a species can be done by adjusting silvicultural interventions in the forest in order to promote more growth space for the selected species, or by implementing forest management regulations that limit access or user rights and specify stakeholder responsibilities. However, forest management regulations and silvicultural systems have their technical, economic and social limits. This is particularly true in those cases and forests where several user groups have conflicting user claims, like for timber, livestock grazing, gathering NWFPs or fuelwood, hunting, or ecotourism. For example, there are often significant value differences among NWFPs, fuelwood and timber for many tree species, and this can lead to conflicts, especially when the benefits from timber and NWFPs go to different forest-user groups. *Entandophragma utile* (sapelli – caterpillars), *Baillonella toxisperma* (moabi – fruits), *Pterocarpus soyauxii* (padouk – resin), *Carapa guianensis* (andiroba – nuts) and *Milicia excelsa* (iroko – bark), to give but a few examples, have both high timber value in overseas markets and high value to the native people who gather NWFPs from them.

Intensified management of nonwood forest product species

Farming NWFP species is ultimately the most viable option to address resource supply constraints when trade demands occur. Domestication of animals and plants began some 10 000 to 12 000 years ago (in the Near East) and this process is still continuing all over the world, particularly in the tropical regions. Several examples of 'new' crops recently introduced from Amazonian species are star fruit (*Averrhoa carambola*), cupuacu (*Theobroma grandiflora*), guarana (*Paullinia cupana*), acerola (*Malpighia glabra*), and pupunha (*Bactris gasipaes*). Animals like the paca (*Agouti paca*) and iguana (*Iguana iguana*) have recently been domesticated in South America.

Moreover, intensified management or farming can also result in better-quality products, more control over the timing and quantity of production, and higher efficiencies in producers' time and resource inputs, while reducing production costs. 'Farming' an NWFP species can also significantly diversify areas of production compared to the limited occurrences of the same species in its natural habitat. For example, the expansion of

walnut trees into many countries, including several in the southern hemisphere, now allows for production during the northern hemisphere winter, making fresh nuts available at world markets almost year-round. Such agricultural expansion spreads the production risks over more producers over a much wider geographical range characterized by many different climatic conditions. If demand levels and prices remain stable over time, rewards for intensifying management will increase. Moreover, having a product available in large and guaranteed quantities from many suppliers worldwide can significantly increase its trade potential.

The process from gathering to domestication is dynamic and complex. It does not always occur in a linear progression in time and space and, indeed, gathering and farming of the same NWFP species can occur at the same time; the species that may be gathered by forest-dependent peoples from the wild in one country may be in production by farmers elsewhere. A good example is the sapota fruit (*Pouteria sapota*) that is still gathered in the Amazon for subsistence and local markets, while at the same time it is being cultivated in California (see the California Rare Fruit Growers at <http://www.crfg.org> and also the Alternative Field Crop Manual, Purdue University: <http://www.hort.purdue.edu/newcrop/afcm>). This can also occur in close geographical proximity, like for industrial plantation rubber and 'jungle rubber' in Indonesia. Introduced crop species may also become invasive in some regions and become a resource for gathering NWFPs in those regions. Examples are *Prosopis* spp. in Africa and *Opuntia* spp. in the Mediterranean, where both are now important for fodder. In contrast, there are instances when a crop falls from favour, farmers stop producing it, and the species reverts to its wild status. One example is the common medlar tree (*Mespilus germanicus*). Originally from the Balkans, it was spread all over Europe where, until the late Middle Ages, it was a popular cultivated fruit tree. In Europe, it is now a 'forgotten' crop, but trees can still be found growing 'wild' in forests.

Nevertheless, domestication and farming NWFP species are not always technically possible, economically feasible or socially and environmentally acceptable; it works well for some species but not for others, like some highly valued mushrooms, mosses or lichens that we do not yet know how to cultivate. Some farmed products may also be qualitatively inferior when compared to wild-gathered products, like those from some medicinal plants. And the economic feasibility of farming NWFPs will be limited so long as the naturally occurring species are widely available and can be obtained at lower prices, e.g., Brazil nuts in the Amazon.

The socioeconomic dimension of domestication is also an important factor. Forest-dependent peoples or socially disadvantaged groups who depend on NWFPs for subsistence and monetary income may not have access to farm land or be able to compete with large-scale production on well-established farms.

Farming NWFPs has important environmental implications in the sense that it can reduce incentives to conserve the ecosystems in which the NWFP species grow naturally. On the other hand, conserving forest biodiversity is a priority because many crops still have wild relatives growing in forests and these wild relatives are a valuable source of genes for plant breeders searching for disease resistant or more productive varieties of crops and domesticated animals.

Gathering NWFPs in forests is felt by some environmental conservation organizations to be more compatible with biodiversity conservation than timber extraction. In reality, this premise depends very much on the type of product and on the way it is harvested. Low density NWFP extraction from natural forests, as occurs for some fruits, leaves or nuts, can have minimal impact on local biodiversity at landscape and species levels. But as harvesting intensity increases and techniques become more destructive, such as killing individual plants to harvest their products, the exploitation of NWFPs can become as harmful to the long-term survival of a species and its related ecosystem as timber extraction. Intensively managed NWFP production systems can even completely displace natural vegetation, like in the case of bamboo shoot production in China.

The long-term effects of promoting commercialization of NWFPs for poverty alleviation and forest biodiversity conservation need further study (see Box 6). If the

Box 6. Can forests be managed sustainably to supply NWFPs for trade?

Since the United Nations Conference on Environment and Development (UNCED) in Rio de Janeiro, Brazil in 1992, NWFPs have increasingly been described as a category of plants and animals with potential to generate income for forest-dwelling people while, at the same time, the associated forests could be included in biodiversity conservation programmes. For example, issues related to the sustainable development of NWFPs were topics of discussion at the World Forestry Congress (WFC) side event *Strengthening global partnerships to advance sustainable development of nonwood forest products* held in Quebec, Canada on 20 September 2003 (<http://www.sfp.forprod.vt.edu/discussion>). This full-day event was co-organized by the International Union of Forestry Research Organizations (IUFRO, Group 5.11 Nonwood Forest Products), the Centre for International Forestry Research (CIFOR), and the Nonwood Forest Products Programme of the Food and Agriculture Organization (FAO) of the United Nations. The meeting identified and prioritized emerging issues for the development of the NWFP sector, drew the attention of the WFC and forest resource decision-makers to key NWFP research needs and policy recommendations for the years ahead, and underlined the relevance of partnership-building among agencies toward creating a sustainable NWFP sector. The recommendations made at the meeting were complemented by discussion summaries and documents on the themes *Commercialization: A reality check*; *Linking NWFP management with livelihood development*; and *Institutional and policy dimensions*. These texts are available online at <http://www.sfp.forprod.vt.edu>.

The meeting presented a wide range of cases of forest management for NWFPs. Some studies showed how NWFP collection significantly improved the livelihoods of forest dwellers without compromising forest biodiversity, while other cases revealed only a modest potential for income generation with high risks of extinction of the harvested species. In some cases it seemed particularly difficult even to make an assessment of the impact of NWFP use on sound forest management and/or on the livelihoods of local communities. This was partly due to the methodological difficulties encountered in carrying out biometric inventories and economic assessments, and to the lack of a consistent terminology to define terms, products and processes related to biodiversity, sustainability or social equity that would allow full comparability of datasets and case studies. Most of the cases reviewed at the meeting showed NWFPs to be supplementing revenue from timber rather than replacing it.

promoted NWFPs become more successful in international trade, demand will increase and, in the long run, their supply will increasingly come from farming, and this will benefit the farmers more than the gatherers and may even result in forest clearing to grow the NWFPs. Moreover, for some domesticated NWFPs, like nut and fruit species, their products will often be larger and of better quality, and they can be supplied with more regularity. In combination, these 'domesticated' attributes of NWFPs can result in their 'forest cousins' completely losing their marketability.

The discussions at the World Forestry Congress (WFC; Box 6) highlighted the need for a multidisciplinary partnership approach for the development of successful livelihood strategies based on the use of NWFPs. Establishing public–private sector partnerships, including those with stakeholders from outside the forestry sector, such as in agriculture, education, industry, trade, tourism and healthcare, were described as essential first steps.

5. Challenges in monitoring the sustainable extraction of nonwood forest products in forests

NWFPs that are supplied from 'farmed' origins are not covered in this section, as the sustainability of their production is no longer a significant issue. In contrast, for those wild NWFPs that are still gathered from forests, major challenges remain in defining sustainable use levels and in developing appropriate management and harvesting methodologies.

Accurate information is needed on the growth and regenerative capacity of those species providing NWFPs (FAO 2001b). Although there is often considerable indigenous knowledge on NWFP species, it is usually geared towards subsistence uses and, as such, it is less applicable to the modern management and marketing needed to sustain production levels for trade. Moreover, it is important to recognize that not all traditional knowledge is biologically accurate. The notion that because it is traditional it must be good needs to be reconciled with scientific data, although scientific knowledge about many NWFPs is poor. Indeed, little scientific effort has been expended to date on sustainable NWFP yields. In fact, very few species providing NWFPs have as yet benefited from targeted research efforts, and basic information about their ecology, growth rates, silviculture, or responses to harvesting is often lacking entirely.

Key biological, socioeconomic and cultural characteristics of NWFPs which cause major challenges for their sustainable management are listed below.

Biological challenges

- Plants and animals that are known resource species for NWFPs are extremely diverse (trees, shrubs, animals, mushrooms, lichens, caterpillars, birds...).
- Many of these species (mushrooms, insects, lianas...) have complex life cycles and population dynamics.
- Different parts from the same plant or animal can be used, with each part requiring different harvesting techniques that ideally should not affect the plant's vigour or reproductive capacity (e.g., trees providing fuelwood, fodder, bark fibres, fruits...).
- Harvesting techniques and intensities depend on the types of product extracted (exudates, leaves, flowers, shoots, fruits and nuts, removal of whole plant...) and on the biology of the species.
- The seasonality of many NWFP species can cause difficulties in detecting them, for example, mushrooms, insects or small animals.

Socioeconomic and cultural challenges

- Many NWFP species have a limited geographical distribution, therefore the products are only of local importance.
- Little to no socioeconomic data are available that quantify the importance of the NWFP to support forest policy and decision-making.
- Many NWFPs are produced in small quantities, generating modest incomes. They may thus be of low priority for professional resource managers or investors.
- Some NWFPs are gathered and used by a range of forest-user groups, often with conflicting interests and needs.
- Many scale differences exist for assessing forest resources, from relatively simple tree inventories to very costly and complex biodiversity assessments which may discourage forest resource managers from including NWFPs in their forest management plans. Time, money and the trained personnel to carry out NWFP assessments are limited.
- Many scientific disciplines, development agencies and conservation organizations include NWFPs (or NWFP-related issues) in their applied research, development or conservation agendas, often with little coordination.
- There is a broad range of different cultural perceptions governing the use of NWFPs around the world, particularly for their use as foods (eating insects, for example).

6. Priorities for developing the nonwood forest product sector

Priorities for the development of NWFP production are grouped here into four clusters:

- 1. Improve the technology for managing NWFP resources in forests.** Low-cost technology solutions for inventorying resources, developing sustainable harvesting techniques, defining harvest levels, and modifying silvicultural regimes to simultaneously obtain timber, forest services and NWFPs need to be devised. Most NWFPs are still harvested from natural systems, so *in situ* sources still dominate NWFP supplies. Existing silvicultural treatments in forests still emphasize timber production, and little attention has thus far been given to the silvicultural needs of NWFP species.
- 2. Better integrate ex situ and in situ production of NWFPs.** How, where and when domestication becomes appropriate as a complementary or substitute strategy for producing NWFPs is a thorny issue. Another is whether domestication, if and when it occurs, will improve the livelihoods of forest dwellers. Producing NWFPs through agriculture (*ex situ* or on-farm production) is a growing solution being driven by market demand, but gatherers are not necessarily the beneficiaries. While domestication may be a valid response to supply shortages, substantial research is still needed to solve the technical issues in both domesticating NWFP species and in identifying the circumstances under which domestication programmes can lead to successful outcomes for forest dwellers.
- 3. Improve social equity in producing, using and trading NWFPs in a 'globalizing' world.** How can forest management become more socially equitable, particularly for the rural communities that depend more on the nonwood resources of the forests? Other challenges are how to reconcile the objectives of different forest-user groups of timber and nonwood resources, how to manage user conflicts and how to have the intellectual property or other rights of NWFP producers adequately recognized and compensated by global trade. For instance, the shift from subsistence use to global commerce for some NWFPs is built directly on indigenous knowledge of these products and their longstanding traditional use. Local communities must benefit from their often very active role in conserving forest resources and from any contributions they make to the development of commercially interesting products. What is the impact of growing trade globalization on NWFP producers? How can entrepreneurship and the development of small enterprises be encouraged in forest communities still operating within village- or local-market contexts?

The roles and impact of non-tariff, trade-related instruments such as certification schemes and best practice codes needs further clarification. The methods currently in place to assess the socioeconomic importance of the NWFP sector at international, national and local levels must be improved in order to address the issues mentioned above.

- 4. Strengthen institutional support for the NWFP sector.** Although informal arrangements to harvest, use and trade NWFPs may in some cases be well-established, in general there are few formal institutional arrangements in place to monitor and regulate the flow of NWFPs from producers to consumers. Formal institutional arrangements based on coordinated multi-agency approaches are needed to address the management and conservation of NWFPs. This is because in many countries forestry, agriculture, environment and/or health ministries deal with different aspects of NWFPs. For example, socioeconomic issues of NWFP production might be covered by ministries or agencies ranging from trade, industry, education, or research. Ministries that are concerned with social issues, such as regulating access to forests, may also deal with land reform or rural development issues, and include NWFPs. This fragmentation of competencies can result in poor management owing to poor communication and poorly coordinated action. Thus, communication among institutions within countries, and synergies among international partners,

must be substantially improved. Policies generated outside forestry sectors may be as important as NWFP policy within the forestry sectors, and these must be included in the development of institutional arrangements governing NWFPs.

7. Conclusions

NWFPs are significant at the local level mainly for subsistence and, when traded, for village-level markets. Their commercial importance at national and international levels is difficult to assess, but is undervalued by the present reporting schemes. The terminology related to NWFPs is ambiguous, and it includes a wide range of terms with different content and scope that makes reporting and information exchange difficult.

The contribution of NWFPs to poverty alleviation needs further investigation. Trade at national and international levels requires large capital investment and social organization usually not available within groups of NWFP producers. Hence, NWFP gatherers can be easily marginalized in large-scale commercial situations.

The impact of gathering NWFPs on the biological diversity of forest species is poorly understood, although subsistence use of NWFPs will in most cases not result in resource depletion. While harvesting NWFPs is generally felt by forestry experts to be less destructive of forests than logging, the commercial exploitation of NWFPs for (inter)national trade is usually undertaken through open-access systems involving uncontrolled quantities that may be unsustainable.

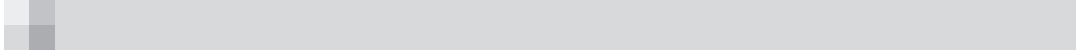
For NWFPs traded on international markets, resource supplies are increasingly being derived from on-farm cultivation. But NWFP domestication appears to be providing more benefits to farmers than to gatherers, though it may also lead to reducing gathering pressure on wild sources of NWFP species. Although this may benefit the conservation of the forest ecosystems, it may lower the value of the wild NWFPs species in forests to local users.

Many challenges remain in defining sustainable use levels and sound practices for managing, harvesting and trading NWFPs. A multidisciplinary approach based on partnerships among NWFP stakeholders and agencies within and beyond the forestry sector is essential if livelihood improvement is the goal.

References

- Belcher, B.M. and M. Ruiz-Pérez. 2001. An international comparison of cases of forest product development: Overview, description and data requirements. Working Paper no. 23, Centre for International Forestry Research (CIFOR), Bogor, Indonesia.
- FAO (Food and Agriculture Organization of the United Nations). 2001a. Global Forest Resources Assessment 2000. FAO Forestry Paper 140. FAO, Rome, Italy.
- FAO (Food and Agriculture Organization of the United Nations). 2001b. Resource assessment of nonwood forest products, experience and biometric principles. Nonwood Forest Product (NWFP) Series No. 13. FAO, Rome, Italy.
- IBGE (Instituto Brasileiro de Geografia e Estatística) – Anuario Estatística do Brasil. 1972, 1982, 2000 and 2003. Diretoria de Pesquisas, Departamento de Agropecuária, Produção da Extração Vegetal e da Silvicultura, Brazil.
- Purnama, B.M., H. Prahastb and B.D. Nasendi. 1998. Rattan in east and south Kalimantan, Indonesia: A case study of the production-to-consumption systems. International Network for Bamboo and Rattan (INBAR) Working Paper no 21, INBAR, Beijing, China.
- Ruiz-Pérez, M., B. Belcher, R. Achdiawan, M. Alexiades, C. Aubertin, J. Caballero, B. Campbell, C. Clement, T. Cunningham, A. Fantini, H. de Foresta, C. García Fernández, K.H. Gautam, P. Hersch Martínez, W. de Jong, K. Kusters, M.G. Kutty, C. López, M. Fu, M.A. Martínez Alfaro, T.R. Nair, O. Ndoeye, R. Ocampo, N. Rai, M. Ricker, K. Schreckenberger, S. Shackleton, P. Shanley, T. Sunderland and Y. Youn. 2004. Markets drive the specialization strategies of forest peoples. *Ecol. Soc.* 9(2):4.

- Sastry, C.B. 2001. Rattan in the twenty-first century – an overview. *Unasylva*, No. 205. FAO (Food and Agriculture Organization of the United Nations), Rome, Italy.
- Stiftung Ökologie and Landbau. 2003. http://www.soel.de/publikationen/oel/oel_inhalt2003.html. Germany.
- Vantomme, P. and S. Walter. 2003. Opportunities and challenges for nonwood forest products certification. *In* Congress Proceedings XII, World Forestry Congress (WFC), 21–28 September 2003. WFC, Quebec, Canada.
- World Forestry Congress(WFC) side event. 2003. Strengthening global partnerships to advance sustainable development of nonwood forest products, held in Quebec, Canada on 20 September 2003 (<http://www.sfp.forprod.vt.edu/discussion>).
- Zitzmann, G. 1999. Multiple use and livelihood strategies in Mopane woodland. The case of Ditladi, northeast district, Botswana. *Schriftenreihe des Institutes für Internationale Forst- und Holzwirtschaft* No. 2. TU (Technische Universität), Dresden, Germany.



Chapter 4

Modelling the biological processes: from genes to ecosystems

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1. Introduction: how can models help us understand biological processes and identify management options?

During the last three decades, simulation models have been used more and more to study forest ecosystems (Johnsen *et al.* 2001; Porté and Bartelink 2002). Models have now become important research and management tools because (i) trees are characterized by long generation times that limit empirical observation of their life cycles; (ii) the complexity of forest ecosystems can be studied to advantage using systems analysis approaches (Berg and Kuhlmann 1993); (iii) as pressures on forests grow, predictions are increasingly needed in forest policy and management decision-making; and (iv) computer and software performance has reached a level of sophistication that generates confidence in simulated forest management outcomes. Models have been used on natural forest ecosystems, on forest plantations and on forests disturbed by humans.

Models have been used to predict growth and yield for forest management and to understand forest demography, dynamics and successional processes. They have been used to predict and analyze the flow and allocation of matter, energy and genetic information and to define new experiments. And they have been employed to estimate the effects of natural and human-influenced disturbances such as fires, pests, storms, air pollution, exploitation and climate change on forest ecosystems.

Forest-related models have been classified in different ways. One broadly used classification distinguishes 'empirical' from 'mechanistic' models. Models are empirical when equations are statistically fitted to data. In contrast, mechanistic models attempt to integrate causal knowledge of physical, biological or mechanistic processes, that is, they attempt to establish functional relationships among elements of modelled systems. Another way to classify models is based on spatial scales, which are typically set at 'stand', 'gap' or 'tree' levels. In stand models, individual trees are not described, and horizontally homogeneous leaf layers represent the canopy. Gap models define and track individual trees competing and growing in spatially restricted areas called 'gaps' that in most forest models range from 10 x 10 m to 100 x 100 m. Other types of models are constructed according to the structure and floristic complexity of the forest to be modelled, such as 'mono-species and even-aged forest' or 'mixed-species and uneven-aged forest', or according to the way the spatial position of trees and competition is determined, such as 'distant dependent' versus 'distant independent' models (distant dependent tree models analyze a given tree's neighbourhood to estimate its competition according to the number and the size of other trees that occur within a certain distance).

Finally, some models have been developed solely for scientific research or principally for management purposes.

The current trend in model development is towards mechanistic models with high spatial resolution (tree or gap models) that integrate processes from different disciplines. An example would be a model that links soil and climatic data with eco-physiological processes of trees (Lindner *et al.* 2002). Nevertheless, empirical models, like those that are applied to traditional forest growth and yield, are still commonly used. Assuming that site conditions remain constant, such models predict future growth from actual measurements of historic growth. Because they ignore potential changes in environment, genetics and silvicultural treatments that may occur between forest rotations (Johnsen *et al.* 2001), they usually cannot be applied to other regions without re-parameterization (a parameter is a numerical quantity that mediates the relationship between variables in a model). However, they do provide accurate predictions when site conditions remain unchanged from the time when the parameters of the model were set. In contrast, mechanistic models have the advantage of being applicable to a broad range of forests, but their predictions (e.g., of growth and yield) are less precise, and the amount of data needed to make them work well is greater than for empirical models, thus rendering them less appropriate for forest management. Currently, most models of forest ecosystems are 'hybrids' containing both mechanistic and empirical elements.

2. Modelling at landscape levels: forest ecosystems

2.1 Forest growth and yield models

One of the oldest and most common uses of models in forestry is in the development and application of yield tables and growth models. During their history, which extends back more than 250 years, yield tables have evolved from elementary versions that used limited datasets for even-aged, mono-specific forest stands to sophisticated computer simulation models that now predict the dynamics of uneven-aged, mixed forests (Pretzsch 2000). The objectives of these models are primarily to predict forest growth and yield, estimate the effects of thinning and logging, and develop guidelines for the management of multispecies forests. For illustration, we will briefly describe two examples of this type of model. Both examples are empirical distance-dependent tree models. The SILVA model was developed for temperate forests in Germany (Pretzsch *et al.* 2002) and the SYMFOR model was developed for tropical forests in Asia and South America (Phillips *et al.* 2003; Phillips *et al.* 2004a).

SILVA

SILVA is an empirical distance-dependent single-tree forest growth model. The model simulates competition, mortality, growth, thinning and harvesting. The results provide classical tree and stand information for forest management, information on timber grading and monetary yield, and statistics on biodiversity and forest structure. SILVA has been applied in forest management for operational and strategic planning, and it has helped in the development of management guidelines for designated tree species or stand types under known site conditions. A typical application is to compare different approaches to forest thinning with timber production and monetary yield (Figure 1). The underlying equations of SILVA were developed from more than 155 000 tree observations made at permanent sample plots in Germany that included Norway spruce (*Picea abies* (L.) Karst.), Silver fir (*Abies alba* Mill.), Scots pine (*Pinus sylvestris* L.), Common beech (*Fagus sylvatica* L.) and Sessile oak (*Quercus petraea* [Mattuschka] Liebl.). In order to start the simulations, the model needs spatial position, diameter, height and species data for all the trees in the stand. The model also includes site differences using parameters for temperature, soil water retention capacity, precipitation, soil nutrient supply, and NO_x and CO₂ concentrations.

As is typical of empirical models, the predictions are valid so long as the future site conditions remain within the range of conditions that were found at the permanent sample plots used to parameterize the model.

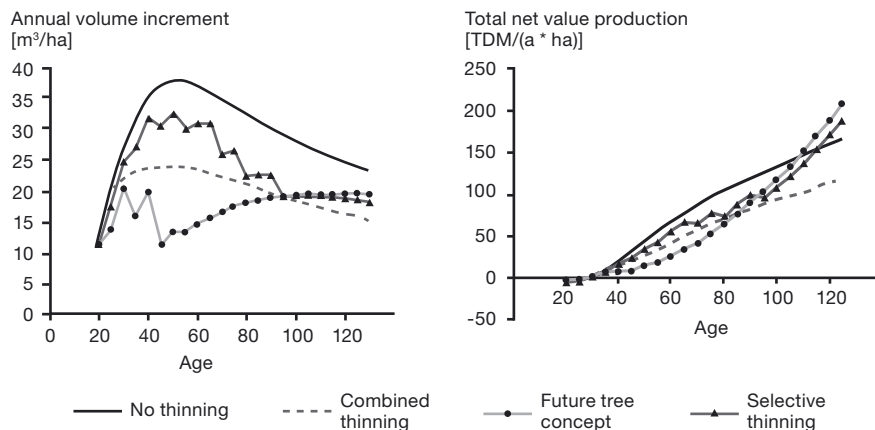


Figure 1. Results using the simulation model SILVA. Simulated annual volume increment (left) and total net value production (right) of differently treated Norway spruce stands in southern Germany. Future tree concept: a selection is carried out by the model. It recommends leaving a total of 250 trees, around which 2–3 competing individuals (on average 2.5 competitors per tree) are removed, at stand top heights of 10, 14, 18 and 22 m. Source: Pretzsch *et al.* 2002.

SYMFOR

SYMFOR is a simulation framework that combines models of natural tropical forest ecology with models that describe important elements of commonly recommended selective logging treatments in tropical forests. The part of the SYMFOR model that concerns natural forest ecology describes tree growth, mortality and seedling recruitment for each tree above a minimum diameter. This kind of model is individual based, meaning that each tree and the unique conditions of its environment are considered separately, and may be spatially explicit. Spatially explicit simulations in SYMFOR use information on the position of each tree to calculate competition through an analysis of its neighbourhood. Forest management measures like the construction of skid trails (temporary pathways used to shuttle logs and trees out of the woods) are given spatial dimensions (known as polygons) so that the damage to trees with known spatial positions can be computed. SYMFOR simulates the conditions that create the state of the forest, rather than simulating the forest state itself, so it is possible to simulate many forest management scenarios that were not described in the initial data used to calibrate the model. SYMFOR has been applied to tropical forests in Indonesia, Guyana and the eastern Amazon in Brazil. The Brazilian version of SYMFOR is based on measurements of all trees with a diameter of 5 cm or more located in experimental plots in the Jarí Forest and the Tapajós National Forest (Pàrà State, Brazil) over a 16-year period. As with most tropical growth models, tree species are grouped according to their recruitment, growth and mortality. For the Amazon version of SYMFOR, ten species groups were used to describe the natural processes affecting tree behaviour. Growth rates were calculated for each species group using the tree diameter and a competition index. The competition index is a numerical value indicating the competition pressure on a given tree. Different equations using information on the number and size of competing trees in a given neighbourhood around a tree, or the relation of the size of a given tree compared to the size distribution of all trees, are applied in forest growth models to calculate this value. Mortality and recruitment are simulated as stochastic processes. Recruitment probability is based on the predicted growth rate of a hypothetical tree. Options exist to vary human interaction with the forest in ways that reflect potential forest management decisions (Figure 2, overleaf).

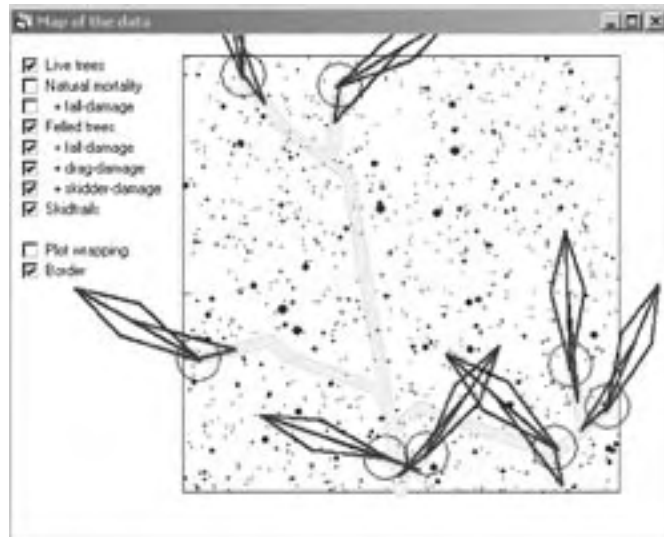


Figure 2. Remaining trees (small dots), predicted fall (lozenges), drag and skidder damage (thick lines and open circles) after a simulated logging operation, using SYMFOR in the Brazilian Amazon.

SYMFOR was applied in the Brazilian Amazon to simulate current forest management practices based on 40 m³/ha of timber extracted with a cutting cycle of 30 years. Results showed that yields could be sustained for three harvests following the first logging of primary forest, but that the composition of timber would move progressively towards lightwood rather than hardwood species (Phillips *et al.* 2004a).

2.2 Mechanistic models to simulate the effects of air pollution and global climate change on forest stands

Many mechanistic models have been developed to understand the reaction of forest ecosystems to stress. Schwalm and Ek (2001) reviewed 12 mechanistic individual tree growth simulators designed to address climate change and air pollution. They argue that models seeking to address climate change and air pollution must include a specific suite of site characteristics. A 'site' is defined as the totality of abiotic and biotic factors that affect tree growth. The types of models reviewed vary from 'stand-level, single species, even-aged' models (Landsberg and Waring 1997) to 'tree-level, multispecies, multi-aged' models (Nikolov and Fox 1994).

Mechanistic models contain various submodels that simulate below-ground processes like nutrient and water cycles, as well as above-ground processes like interactions with the atmosphere and other physiological processes. Temperature, precipitation and wind speed are integrated into these models as external variables that influence stomatal conductance, photosynthesis and hydrology. The length of the simulated time steps of the variables can fluctuate from model to model. Despite its importance in climate change, carbon dioxide concentration and its impact on photosynthesis are only included in a few forest models (Chen *et al.* 1998). The direct impact of atmospheric depositions on trees is less well understood than their indirect effects on soil conditions and forest nutrient cycling. Thus, mechanistic models are more sophisticated in simulating below-ground processes. For example, several models accommodate atmospheric inputs into nutrient pools with varying degrees of detail and data requirements, and the mechanistic models TREEDYN3 (Bossel 1996) and FORSANA (Grote and Erhard 1999) employ modifiers to

depress photosynthesis. Other models treat only NO_x and use it as an addition to the plant nitrogen pool. More sophisticated simulations on the impact of pollutants are possible with the model ACIDIC (Kareinen *et al.* 1998). This model includes a soil chemistry sub-model that can include daily atmospheric depositions and their effects on nitrogen cycles, carbon dioxide dissolution, aluminium complexation, and several other soil chemical processes in a multilayered soil column.

Other features included by mechanistic models are soil temperature, moisture status, microbial activity, nutrient availability, photosynthesis, crown geometry and the mechanisms of light interception.

In multispecies models, species-specific global mean values are given for plant relevant parameters such as growth rates and photosynthesis. However, most process-oriented mechanistic forest ecosystem models do not include genetic variation within species. An exception is the integration of phenological variation for different tree provenances by Berninger (1997). This is a branch of forest genetics specializing in so-called 'provenance tests'. These are field experiments or controlled condition experiments (e.g., in greenhouses) that study the genetic bases of phenotypic variation (e.g., growth, eco-physiological parameters, phenology or resistance to pests) in trees with different geographic origins (see Section 3.3).

2.3 Contributions of geographic information systems (GIS) and remote sensing applications to forest ecosystem modelling

Many of the models mentioned thus far contain spatial components, thereby recognizing that forest species and their habitats are neither homogeneous nor randomly distributed across landscapes. The spatial distribution and arrangement of trees play important roles in ecosystem processes, and the distribution and density of tree species in forests often vary according to soil type, water regime, light availability and climate. Spatial variation in forest-related features occurs on many scales, from the microlevel to the macrolevel, and this variability can play a fundamental role in determining the nature and magnitude of certain ecosystem processes. A brief review of some applications of GIS and remote sensing to forest ecosystem modelling is presented here.

Remotely sensed data such as satellite images or aerial photos of forests, or GIS datasets of associated environmental variables like climate, topography or soil, can provide a wealth of information about the spatial arrangement of important forest features in models. They also play a substantial part in constructing and calibrating models. Moreover, information like this, and especially that derived from remotely sensed images, can be used in validating model performance. Remotely sensed data can also reduce the cost of time-consuming field measurements, while at the same time providing comprehensive datasets that cover very large areas.

Because forest spatial features are so ubiquitous and important to forest structure and function, GIS and remote sensing technologies are now playing important parts in forest ecosystem modelling, whether through the software used for modelling spatial processes or by providing associated spatial data. Off-the-shelf commercial GIS and remote sensing packages like ESRI's ArcGIS or ERDAS's Imagine are not specifically designed for spatiotemporal modelling, focusing more on generic data manipulation and spatial analysis. Other software packages like ArcInfo (also from ESRI) or the PCRaster environmental modelling language (from Utrecht University, the Netherlands) provide better performance and flexibility in spatiotemporal modelling of forests by combining advanced spatial pattern analysis with temporal processes.

Simple forest cover is the most basic forest feature to be monitored using remote sensing. Landsat satellites in particular have been used to monitor changes in forest cover (Skole and Tucker 1993; Helmer *et al.* 2000; Pax-Lenney *et al.* 2001). For instance, changes in reflectance of Landsat bands are noted when forest is cut, with bare soil reflecting considerably more light. The Advanced Very High Resolution Radiometer (AVHRR) satellite-mounted sensor has provided numerous 1-km resolution datasets of land use from around

the world. This sensor is often able to distinguish forest types ranging from evergreen to deciduous and broad-leaved to needle-leaved, as well as canopy density (Belward 1996; Forest Resource Assessment 2000). The SPOT satellite-mounted sensor has also produced 1-km datasets (Global Land Cover 2000, <http://www.gvm.jrc.it/glc2000>). Foody and Hill (1996) introduced a method for classifying 12 tropical forest types using Landsat data.

Studies that require finer scales within a forest stand or canopy can also use remote sensing to acquire data on structural and ecological features of forests. Woodcock *et al.* (2001) have established methods for monitoring forest change over time using Landsat images. They were able to identify logging operations as well as defoliation caused by insect damage in a temperate forest with Landsat. Similarly, Michalek *et al.* (2000) used Landsat images to identify fire severity and stand density in an Alaskan spruce forest. Several studies have been able to identify phenological cycles employing remotely sensed data (Everitt and Judd 1989), often using changed phenological patterns over time to distinguish among forest species (Thomasson *et al.* 1994; Wolter *et al.* 1995; Key *et al.* 2001). The forestry community has also used aerial surveys to monitor tree positions and to manage plantations in temperate environments. Pouliot *et al.* (2002) have pioneered a method for detecting individual tree crowns using high-resolution aerial imagery of coniferous forests, thus making it possible to calculate structural parameters of individual trees on large scales and to monitor tree growth over time. Wasseige and Defourny (2002) use multiple SPOT images of tropical forests to assess such structural parameters as canopy roughness or the presence of gaps through bi-directional reflectance analysis. Successional stage and/or succession over time has also been monitored in both tropical (Helmer *et al.* 2000) and temperate forests (Fiorella and Ripple 1993) using satellite imagery. With the advent of higher spatial and spectral resolution imagery, detailed structural parameters of trees may eventually become quantifiable (Asner *et al.* 2002).

This review has presented several examples of remote sensing technology and how it has been applied to monitor and to model forest ecosystems. As we have seen, among their many applications satellite images have been used to monitor forest cover, forest structure and successional processes. The principal benefit of integrating remotely sensed data into forest models is that large datasets over large areas can be generated using nondestructive and comparatively rapid methods.

3. Modelling at the gene level: forest genetic resources

We will show in the following short overview that the genetic structure of tree populations results from complex genetic systems that include processes and mechanisms of organization, expression, reproduction, combination, and temporal and spatial distribution of genetic information (Darlington 1939), both past and present. Different aspects of these genetic systems can be found in all the models used in population genetics.

In comparison to other organisms, trees have a relatively high level of genetic variation (Ledig 1986; Hamrick and Godt 1990; Nybom 2004). Two explanations are usually given to account for this. Firstly, they need it to guarantee high adaptability to diverse environments. This is particularly important for sessile trees with long life spans. Secondly, because of their long life spans, large numbers of individuals and overlapping generations of populations, trees accumulate mutations that maintain genetic variation. Owing to its importance for adaptation and adaptability, the genetic diversity of tree populations has a significant function in ecosystem stability (Gregorius 1991). Thus, information on the structure and dynamics of genetic variation within tree populations is important in efforts to conserve forest genetic resources.

Various processes affect the genetic structure and population dynamics of trees. Mutations generate variants of the same gene (alleles) and are the unique source of new genetic variation, though the frequency of mutations is often low. Mutation rates of 10^{-6} per gene locus per generation have been reported for plants. Hence, it is clear that genetic variation resulting from mutations can only be generated over very long periods (Gill *et al.* 1995).

Genotypes are formed from combinations of alleles and, depending on the mating system, these alleles can be differentially arranged into genotypes during reproduction. This includes all the processes combining haplotypes in zygotes (e.g., sexual types, self-incompatibility systems, flowering phenology and spatial distribution of pollen). The dispersal of seeds within and among populations (= seed migration), viability selection and random genetic changes due to population size reductions (genetic drift) determine the genetic composition and spatial genetic structure in the resulting ontogenetic stages that range from seeds to reproductive trees. The biotic and nonbiotic environments have an important influence on these processes, as do human activities, such as when seeds are introduced. Forest exploitation and forest management can also change the genetic structure of tree populations (Hosius 1993; Aldrich and Hamrick 1998; Rajora 1999; Takahashi *et al.* 2000). Products of human activity, such as air pollution (Scholz *et al.* 1989), may indirectly influence population genetic processes by affecting the environmental conditions to which forests must adapt. Since it is known that major changes in genetic patterns occurred in temperate and tropical tree populations as a consequence of extinction and recolonization during and after glacial periods (Petit *et al.* 1997; Caron *et al.* 2000), similar effects are expected from ongoing climate change (Giannini and Magnani 1994; Kremer 2000).

3.1 Small-scale or short-term simulations

Systems analysis approaches using simulation models are helpful in analyzing the temporal and spatial dynamics of complex systems (Berg and Kuhlmann 1993). Simulation models have been used in forest genetics to estimate or predict parameters of particular population genetic processes. For example, the mixed mating model fashioned by Ritland and Jain (1981) has been used to estimate the proportion of self-pollination, outcrossing and biparental inbreeding in many tree species (Friedman and Adams 1985; Murawski and Hamrick 1991; Bacilieri *et al.* 1996; Burczyk *et al.* 1996; Rossi *et al.* 1996; Doligez and Joly 1997; Collevatti *et al.* 2001). Hardy and Vekemans (1999) developed a model that estimates gene flow parameters from spatial genetic structures of tree populations. The 'two-generation approach' compares the genetic composition of seeds from different trees. Information on the genotypes of the mother trees is then used to separate haplotypes of seeds coming from the mother tree from haplotypes coming from the pollen donors (father trees). Strong gene flow by pollen leads to uniform genetic compositions of effective pollen clouds, whereas limited gene flow is linked to stronger genetic differentiation of the effective pollen clouds of single tree progenies. The observed genetic differentiation is fitted to theoretically expected values in order to estimate a function of pollen dispersal. Recently, 'two-generation approach' models have been developed from data on genetic differentiation among pollen clouds of single tree progenies to estimate pollen dispersal and effective densities of reproductive trees (Austerlitz and Smouse 2001; Smouse *et al.* 2001; Austerlitz and Smouse 2002; Dick *et al.* 2003; Degen *et al.* 2004).

Great effort has been expended on developing and applying simulation models to the dynamics of spatial genetic patterns within and among tree populations. Doligez *et al.* (1998) analyzed factors that influenced the spatial genetic structure and the level of inbreeding of a tree population by using an 'isolation-by-distance' model (in 'isolation-by-distance', gene flow occurs among local neighbourhoods in a continuously distributed population). The density of trees, distances of pollen and seed dispersal, overlapping of generations, and selection on a biparentally inherited locus with two alleles were included in this simulation. The spatial clustering of trees and the outcrossing rate were found to have an important effect on spatial genetic structure and on the level of individual tree inbreeding. The degree of inbreeding at the population level was mostly predicted by the degree of generational overlap.

Very few demographic models have been able to estimate the genetic impact of human activities on tree populations in a more sophisticated manner (Alvarez-Buylla *et al.* 1996). Nevertheless, a set of models designed to make better simulations of the mating systems and the gene flow of tree generations has been developed by Gömöry (1995). While these models deal well with different levels of allelic diversity and spatial structure, and

are able to simulate different modes of pollen and seed dispersal, important genetic and demographic processes and characteristics such as overlapping generations, flowering phenology and tree growth were not included.

3.2 Long-term, large-scale simulations

Metapop

Sophisticated simulations of the large-scale population dynamics of trees have been made with the simulation model Metapop, which is designed to study the genetic evolution of a subdivided population of a diploid species under natural selection. This model combines population genetics, quantitative genetics and population dynamics. Using Metapop, Le Corre *et al.* (1997) tried to reconstruct the postglacial recolonization of oaks in Europe from different refuges across a 100 x 300 km region by using simulations of a two-dimensional (2D) 'stepping-stone model' in which each population receives migrants from neighbouring populations. The simulations were made for a set of populations connected by migrations of seeds over long time periods (many generations). The authors simulated the distribution and spatial pattern of maternally inherited chloroplast haplotypes and compared them with experimental data. They found that a spatially restricted distribution of acorns combined with rare long-distance seed dispersal events (several tens of km) best explained the speed of the observed recolonization and genetic differentiation among populations.

In another application of Metapop, Le Corre and Kremer (2003) studied genetic variability in a subdivided population of trees under stabilizing and diversifying selection conditions using three indicators: neutral markers, quantitative trait loci (QTLs), and the trait itself. The objective of this simulation was to explain differences in both quantitative and genetic traits among experimental tree populations (Jaramillo-Correa *et al.* 2001). Quantitative trait loci are regions on a gene or genes linked to a particular trait. They have been studied in trees for traits like growth, physical and chemical wood characteristics, bud set and other morphological and phenological characters. A quantitative model with additive effects was used to link genotypes to phenotypes. Using an analytical approach, the authors compared the phenotypic diversity within a deme (H_S) – a locally interbreeding population – to the genetic variance within a deme (V_W), assessed using neutral marker differentiation (F_{ST}) and the quantitative differentiation (Q_{ST}) for the trait. The difference between the results using F_{ST} and Q_{ST} was shown to depend on the relative amounts of co-variance between QTLs within and between demes. Simulations were used to study the effect of selection intensity, variance of optima among demes, and the migration rate for both an allogamous and a predominantly self-pollinating species. Contrasting genetic variability using neutral markers, QTLs, and the trait were observed to be a function of the level of gene flow under diversifying selection conditions. The greatest discrepancy among the three indicators occurred under highly diversifying selection conditions and high gene flow. Thus, the study reveals that diversifying selection conditions might cause substantial heterogeneity among QTLs, as only a few of the markers show allelic differentiation, while the majority behave as neutral markers.

Impact of colonization on genetic diversity and differentiation

Austerlitz *et al.* (2000) developed a model to study the impact of the life cycle of forest trees, and in particular the length of their juvenile phase, on genetic diversity and differentiation during the last glacial period in Europe and the following colonization period. Their 2D stepping-stone model includes demographic structure (matrix model), mutations, pollen and seed dispersal and density-dependent fertilities (Figure 3). The simulation study was motivated by the observation that tree species have high within-population diversity and low between-population differentiation of their nuclear genes. This is in contrast to annual plants which show much more differentiation for nuclear genes but much less overall diversity (Nyblom 2004). The usual explanation for

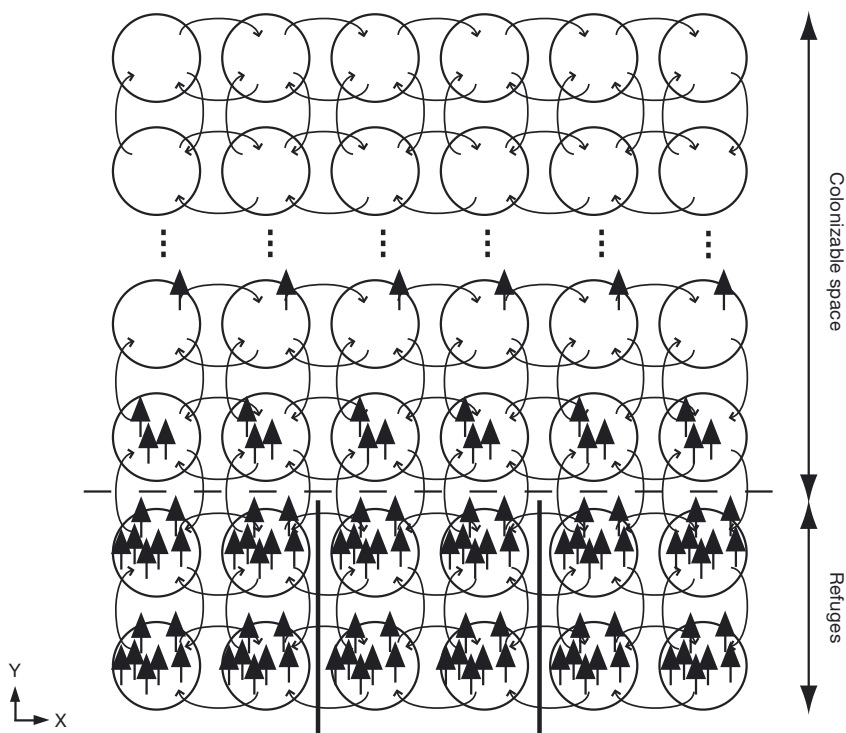


Figure 3. Schema of the two-dimensional stepping-stone model used to simulate the impact of postglacial recolonization on genetic diversity and genetic differentiation among tree populations. Source: Austerlitz *et al.* 2000.

this difference is that pollen flow, and therefore gene flow, is much greater for trees (Liepelt *et al.* 2002; White *et al.* 2002). However, this explanation is problematical because trees have only very recently recolonized temperate areas and have experienced many founder events that usually reduce within-population diversity and increase differentiation among populations (Scotti *et al.* 2000; Kremer *et al.* 2002). Only extremely high levels of gene flow could counterbalance these successive founder effects. Austerlitz *et al.* (2000) showed with their model that both a reasonably high level of pollen flow and a certain demographic distribution of trees were needed to explain the observed low genetic differentiation among most European tree populations. With the help of simulations, the weak founder effect was explained by delayed reproduction of trees during recolonization. When the first trees on the colonization frontier reach reproductive age, a non-negligible part of the space is already occupied by juveniles from seeds that arrived earlier from colonized areas and were stored in the soil seed bank. Furthermore, they showed that gene flow and life cycle also have an impact on maternally inherited cytoplasmic genes. These are characterized, both in trees and annual species, by much less diversity and much more differentiation than nuclear genes.

3.3 Genetic variation and global change

Another category of simulation models used in forest genetic studies attempts to estimate the genetic adaptability of tree populations to global climate change. In most cases, predictions are based on quantitative genetic data from provenance tests, common garden experiments or data from international phenological gardens using clones (Beuker 1994; Kramer 1994;

Matyas 1994; Liesebach *et al.* 1999; Chuine *et al.* 2000). In common garden experiments, plants of different origin (genetically different material) are grown under the same controlled environmental conditions. Such experiments help forest scientists understand the genetic basis of the variation in the studied traits (e.g., annual growth rate, photosynthesis, frost hardiness and resistance against insects). Stimulated by a recommendation from the World Meteorological Organization (WMO), a network of International Phenological Gardens (IPGs) was established in 1957 by the National Hydro-Meteorological Services (NHMSs) of several European countries. The idea underlying the establishment of this network was to observe the behaviour of cloned plants under different climatic conditions throughout Europe (from Finland to Greece and from Ireland to Russia).

Genetically determined variation in *Pinus sylvestris* was included in the process-based model SICA (Simple Canopy Model) by Berninger (1997). The model was applied to a geographic transect at six meteorological stations in Europe. It simulated photosynthesis and transpiration at the leaf layer, and included respiration and soil water balance as functions of canopy properties and the environment. The model accounted for possible genetic adaptation of the phenology of photosynthesis to the local climate and to decreases in gas exchange owing to drought. In terms of photosynthesis, length of growing season and gross primary productivity (GPP), simulations adding genetic adaptation to the local climate differed by up to 20% from simulations neglecting genetic adaptation.

With the objective of analyzing potential changes caused by global warming to interspecific and intraspecific tree diversity, Takenaka (2001) developed an individual-based model of a forest characterized by spatial structure and gene flow. The forest was represented as a 2D lattice (or grid) of 'cells' (1000 x 100), each of which could sustain at most one tree. A temperature regime gradient was placed along the long side of the lattice. Three groups of tree species were considered, with each group consisting of five species. The three groups differed in their suitable temperature regimes and, because temperature conditions affect the fecundity of trees, they also differed in the amount of seeds and pollen produced; the greater the difference between a tree's assumed optimal temperature and the temperature of the cell it occupied, the lower the fecundity. Each tree had a pair of alleles (genes), that determined the optimal temperature for the tree. A new tree inherited these alleles from the parent trees, which died stochastically. A new tree sprouting from a seed randomly chosen from those dispersed from neighbouring mature trees (age \geq 25 years) then occupied the vacated cell. The densities of seeds and pollen dispersed from a mature tree followed an exponential function. Using this model, Takenaka investigated the possible consequences of climate change for the three groups of trees and found that the new distribution of trees following climate change was affected by the similarity between their ideal habitats. The model showed that the more similar the trees' habitat ranges, the less likely the tree distribution pattern was to change. Using the model, Takenaka also found that species diversity was reduced at the same time due to the founder effect, that is, dominant species near the front line also dominated the new area.

Liesebach (2002) used the forest growth model SILVA described above to study the possible effects of increased annual mean temperature and decreased rainfall on the genetic structure of trees. Inventories of six isozyme gene loci in provenance trials of Norway spruce (*Picea abies* [L.] Karst.) were used to estimate the genetic variation and differentiation of the spruce populations. The data were taken from a 25-year old Norway spruce trial located in Germany. A virtual stand of Norway spruce was generated from these data to simulate growth over a period of 75 years under seven environmental conditions, each of which was attributed two mortality scenarios: one where losses were due to natural death through competition, and another characterized by natural mortality combined with thinning from above (trees are removed from the middle and upper portion of the range of crown and diameter classes).

Allele and genotype frequencies were determined for the trees at 100 years. For each simulated environmental condition, various parameters such as genetic diversity and heterozygosity were determined, using projections from the isozyme data. No substantial differences were found within the genetic parameters. Instead, the highest differentiation was found to lie among three growth parameters, which seemed to bear little relationship

to the isozyme markers. It was concluded that changes in genetic structure over time were more likely to be caused by the genetic drift associated with declining population size, than by global warming.

3.4 Estimating the impact of forest fragmentation and forest exploitation on genetic diversity

Studies simulating the impact of forest exploitation, other silvicultural practices and forest fragmentation on genetic diversity of tree populations are uncommon, and those that exist usually contain oversimplified representations of biological processes. Glaubitz *et al.* (2003) used Monte Carlo simulations to investigate whether the differences between two eucalyptus stands with distinct silvicultural histories in Australia could be explained by genetic drift. In other conservation genetic approaches, minimum viable populations have been estimated by combining models of population genetics with estimates of rates at which mutations arise (Alvarez-Buylla *et al.* 1996). In these models, two factors are expected to threaten tree population survival: (i) loss of potential adaptive genetic variation due to genetic drift; and (ii) increased inbreeding depression owing to self-pollinating and mating among relatives. Two disadvantages of these models are that they include strong generalizations that are usually not linked to experimental data, and that they do not consider specific forest management operations.

ECO-GENE

The simulation model ECO-GENE was developed by Degen *et al.* (1996) at the Federal Research Centre for Forestry and Forest Products (BFH) in Germany to study temporal and spatial dynamics of genetic structures in tree populations of temperate forest ecosystems. It is an individual distance-dependent tree model that combines elements of forest population genetics, dynamics, growth and management models. Overlapping or separated generations can be created and different processes like gene flow, mating systems, flowering phenology, selection, random drift and competition can be simulated (Figure 4, overleaf). The ECO-GENE model has been used to study the impacts of different silvicultural practices and of air pollution on the genetic structure of tree populations in temperate forests in Europe (Germany, France and Austria) and Asia (Japan) (Degen and Scholz 1996; Degen *et al.* 1997; Geburek and Mengel 1998; Degen *et al.* 1999a; Takahashi *et al.* 2000; Degen *et al.* 2002). Work has been in progress since 1998 to adapt ECO-GENE for use on tropical tree species, and modules on pollen and seed dispersal by animals and flowering phenology have recently been added (Degen and Roubik 2004). The recent integration of the forest simulation model SYMFOR with ECO-GENE offers improved modelling of growth processes and management impact in tropical forests in South America (Phillips *et al.* 2004b).

ECO-GENE and the new ECO-GENE+SYMFOR model are now being used to estimate the effects of forest operations on genetic diversity of tree populations in several projects in tropical and temperate forests. These include the 'Dendrogene' project in Brazil (<http://www.cpatu.embrapa.br/dendro/index.htm>), the EU INCO project 'Geneotropico' in several South American countries (<http://www.nbu.ac.uk/geneo/>), and the EU project 'Oakflow' (<http://www.pierroton.inra.fr/Oakflow>).

For initialization, the ECO-GENE model requires information on the spatial position, diameter and genotype of all trees within a stand or population. Several plots were established in temperate and tropical forests to collect these data. In northern Germany, position and diameter were inventoried and genotypes were analyzed using allozymes and nuclear microsatellites in two mixed oak and beech stands (Figure 5, on p. 83) (Scholz and Degen 1999). A 500-ha plot in the National Forest of Tapajós in Brazil, established within the framework of the Dendrogene project, gathered data for the initialization and the parameterization of simulations with ECO-GENE (Kanashiro *et al.* 2002). Dendrogene is collaborating with the 'Sustainable Forest Management for Timber' project of the Brazilian Institute for Environment and Natural Renewable Resources (IBAMA) and the International Tropical Timber Organization (ITTO). Genetic, reproductive biology and

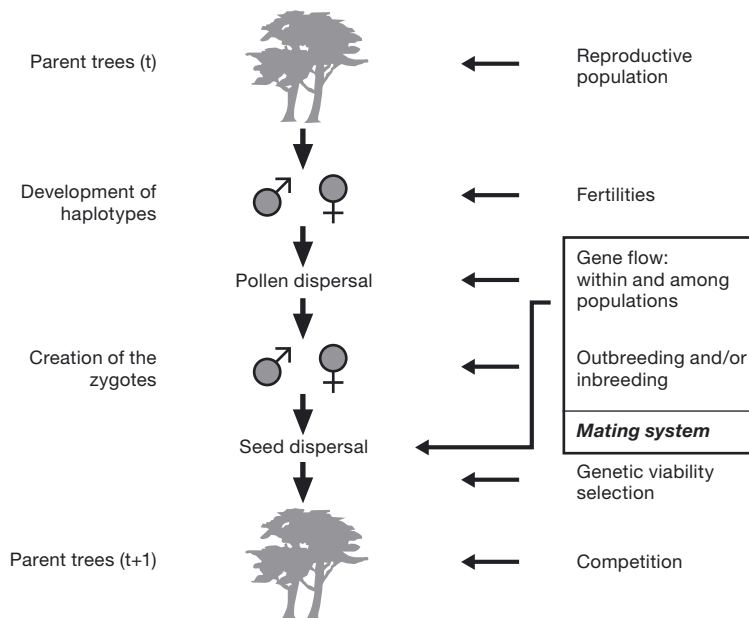


Figure 4. Different processes and factors integrated into the ECO-GENE simulation model. Source: Degen *et al.* 1996.

ecological studies of seven timber species were carried out in Tapajós, before and after logging.

For the parameterization of the different processes (Figure 4), ECO-GENE requires information on pollen and seed dispersal, species phenology, variation in fertility, growth, mortality, selection and self-incompatibility. This information is either elaborated from data gathered in intensively studied plots (ISPs) or is found in the literature. Incomplete data for initialization of the simulations can be filled in by a data generator. The data generator uses aggregated information like allele frequencies and tree densities in diameter classes to create artificial tree populations that can be used to start the simulations. A comparison between simulated and actual allele frequencies in single tree progenies in the ISPs was used to validate simulation outputs.

Example of an ECO-GENE application

Large segments of forest stands are artificially regenerated in several European countries. For regeneration purposes, seeds are usually harvested in certified stands. Different factors such as number and spatial distribution of harvested seed trees, variation in flowering phenology and fertilities of all reproducing trees, pollen dispersal, mating system, and the spatial and demographic structure of the stand determine the genetic composition of the harvested seeds. From a genetic aspect, good seed harvesting results in seeds with a genetic composition that is identical or similar to the original adult population, or which have genetic variation that is equal to or greater than that of the adults.

In order to ensure quality control in regenerating such forests, a sensitivity analysis was carried out with ECO-GENE at the Institute for Forest Genetics and Forest Tree Breeding in Grosshansdorf, Germany. The objectives of this analysis were: (i) to determine the relative importance of the factors cited above on the genetic composition of harvested seeds; (ii) to estimate the risk of genetic erosion due to the seed harvesting procedure; and (iii) to identify thresholds of tolerable genetic differences between the adult population and the

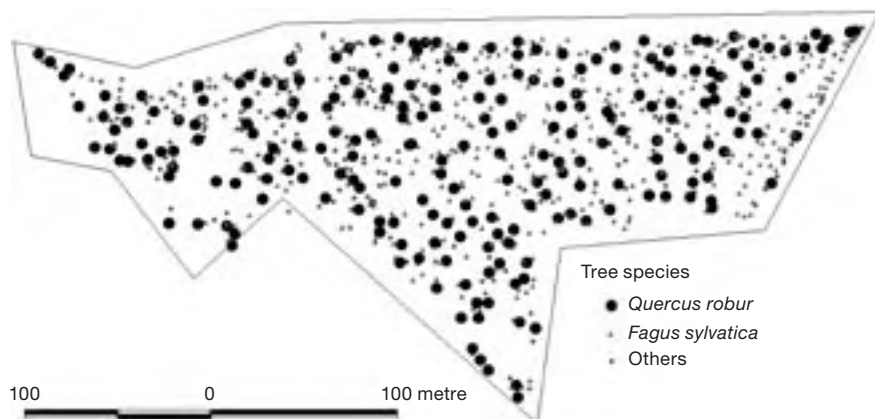


Figure 5. A 5.5 ha ISP (intensely studied plot) established in Behlendorf (northern Germany) to apply ECO-GENE simulations on oak (*Quercus robur*) and beech (*Fagus sylvatica*) populations. All trees with a diameter at breast height (dbh) of 20 cm were measured. Their dbhs, positions and genotypes were inventoried using allozymes and microsatellites.

harvested seeds. The simulation was initialized with the oak dataset from the Behlendorf plot (Figure 5). Information gathered on 229 *Quercus robur* trees included dbhs of ≥ 20 cm, spatial position and the genotypes using seven allozyme and three microsatellite loci. One thousand simulations were run with different parameter configurations selected randomly from within a given range, such as number of seed trees harvested (10–40), coefficient of variation for fertilities ($50 < CV\% < 300$), standard deviation in starting date of flowering (2–7 days), and the exponent of pollen dispersal (0.001–0.05). In each simulation, a sample of 1000 seeds was produced for the selected seed trees, and the response variable was determined as the genetic distance (Gregorius 1978) between all adult trees and the sampled seeds. Finally, the sensitivity of the response parameter to the variation of input parameters was tested with a stepwise multiple regression analysis performed using rank-transformed values (Nathan *et al.* 2001). Results are given in Table 1.

The simulations showed that the genetic distance between the adults and the simulated seeds varied from 0.048 to 0.243. Using the same gene markers, the genetic distance between the Behlendorf *Quercus robur* stand and another stand in the same region was

Table 1. Results of the ECO-GENE sensitivity analysis. Stepwise multiple regression of the genetic distance against the main input parameters of seed harvesting scenarios using the Behlendorf oak dataset.

Genetic distance (adults – seeds) 0.048 < 0.100 < 0.243 (95% \leq 0.147)				
Step	Parameter	R ²	β	P > t
1	Number of seed trees	0.321	-0.564	0.000
2	Variation of fertilities	0.432	0.326	0.000
3	Pollen dispersal	0.479	0.224	0.000
4	Flowering phenology	0.498	0.140	0.000

R² = the cumulative fraction of the variance accounted for by the model, adjusted for the number of independent variables – the ‘importance’ of the parameter; β = standardized regression coefficient – the correlation between the variable and the parameter; P = probability; t = t-statistics.

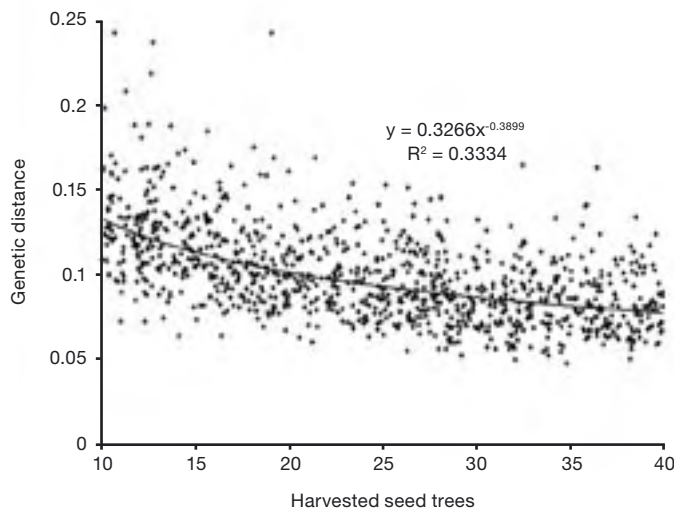


Figure 6. Correlation between the number of harvested seed trees and the genetic distance between adults and seeds within 1000 simulations of a sensitivity analysis on a European oak forest using the ECO-GENE model.

measured as 0.077 (Degen *et al.* 1999b). This value and the 95% confidence interval of the simulations ($95\% \leq 0.147$) can be used to estimate tolerable genetic distances between the original stand and harvested seeds for these same gene markers. In the sensitivity analysis, the number of seed trees was the most important input parameter ($R^2 = 0.32$), and was found to be negatively correlated with the genetic distance. This means that as the number of seed trees increases the genetic difference between the adult population and the seeds will be progressively lower (Figure 6). The variation in fertility was the second most important parameter, and it improved the explained variation of the model up to $R^2 = 0.43$; increasing the variation in fertility results in a higher genetic distance between adults and seeds. The tested parameters of pollen dispersal and flowering phenology had only minor influences on the genetic distances between adults and seeds. These parameters improved the explained variation by only 2% ($R^2 = 0.49$).

The results of the simulations have various practical implications. They can be used to determine critical thresholds for seed harvesting by identifying a minimum number of seed trees, assessing the impact of variation in flowering patterns, or by making available reference values of genetic distances between adults and harvested seeds. Furthermore, the simulations can also carry out sensitivity analyses, which guide research in focusing on important processes and parameters.

4. Research gaps and development issues

A significant problem that exists in efforts to integrate genetic variation and its dynamics into forest ecosystem models is the poor linkage between the genetic variation that is screened by gene markers and the genetic variation that determines differences in adaptive traits. Thus far, ecosystem models have integrated genetic variation as a source of variance of model outputs without knowing the underlying genes and their inheritance (Berninger 1997). Moreover, the dynamics of genetic variation screened by gene markers have been incorporated into genetic models in the absence of related information on their adaptive significance (Liesebach 2002). Understanding the link between genetic variation and its adaptive significance requires the identification of single nucleotide polymorphism (SNP)

markers in coding regions of QTLs and an analysis of genotype x environment interaction for each of the SNPs. These types of studies are now being developed for economically important species (Neale *et al.* 2002), and there is hope that at least some adaptive traits can soon be linked to genotypes. Nevertheless, there is a growing need to improve the integration of quantitative genetic data from provenance and common garden experiments into process-based forest ecosystem models (Berninger 1997).

Several plant species have been studied using both biochemical and molecular gene markers to assess population genetic structure, genetic differentiation among populations and parameters of genetic processes such as pollen and seed dispersal, levels of outcrossing and spatial genetic structure. Depending on the type of gene marker being used, these studies have generated different results. This may be caused either by too few gene loci or individuals sampled, or by natural variation of population genetic processes in the different species studied. There are now more and more molecular gene markers available for population genetic studies in trees. Genetic inventories with amplified fragment-length polymorphisms (AFLPs) offer the possibility of studying more than 100 loci (Vos *et al.* 1995), and the level of variation that can be screened with new markers such as nuclear microsatellites is now high enough to permit the study of gene flow and spatial structure. As a consequence, marker-based problems will become less frequent and less limiting in the near future. In the past, recommendations on sampling strategies for genetic inventories have been elaborated using analytical and numerical models (Gregorius 1980; Krusche and Geburek 1991), and the design of sampling strategies is still an important field for future model applications. For the management of forest genetic resources and for the parameterization of the models, it is essential to know the natural variation of population genetic processes. For this purpose, it would be useful to establish a network of ISPs where these processes could be monitored over several years.

Current forest genetic studies using simulation models allow better estimates of the potential loss of genetic variation as a result of forest exploitation, thinning and seed harvesting than were possible in the past. However, an open issue is the identification of critical levels of genetic variation that do not compromise population survival. What is the minimum threshold of genetic diversity that a certain population must possess to avoid extinction? The same question has been addressed in studying the relationship between species diversity and ecosystem stability (Schwartz *et al.* 2000). Most empirical studies and modelled predictions show saturation of ecosystem function and stability with just a limited sample of local species' variation. This observation supports the conclusion that most plant communities are characterized by the strong dominance of a few species that provide most of the community biomass. Many population genetic inventories have highlighted the existence of so called 'minor polymorphisms' that are characterized by one very frequent and a few rare alleles. This has led to a distinction being made between active and lethal adaptive potential by assuming that the frequent alleles are important for fitness under actual environmental conditions and that the rare alleles might be relevant for future adaptation (Gregorius and Bergmann 1995). The importance of rare alleles for adaptation of a population is a matter of debate (Bush *et al.* 1992). The adaptive relevance of different gene markers and the relationships among levels of genetic diversity, adaptation and adaptability are some of the most important unanswered questions in plant genetics. Modelling will be an important tool in answering these questions.

References

- Aldrich, P.R. and J.L. Hamrick. 1998. Reproductive dominance of pasture trees in a fragmented tropical forest mosaic. *Science* 281:103-105.
- Alvarez-Buylla, E.R., R. Garcia-Barrios, C. Lara-Moreno and M. Martinez-Ramos. 1996. Demographic and genetic models in conservation biology: Applications and perspectives for tropical rain forest tree species. *Annu. Rev. of Ecol. Syst.* 27:387-421.
- Asner, G.P., M. Palace, M. Keller, R. Pereira Jr, J.N. Silva and J.C. Zweede. 2002. Estimating canopy structure in an Amazon forest from laser rangefinder and IKONOS satellite observations. *Biotropica* 34:483-492.

- Austerlitz, F. and P.E. Smouse. 2001. Two-generation analysis of pollen flow across a landscape. II. Relation between $\phi_i(f)$, pollen dispersal and interfemale distance. *Genetics* 157:851-857.
- Austerlitz, F. and P.E. Smouse. 2002. Two-generation analysis of pollen flow across a landscape. IV. Estimating the dispersal parameter. *Genetics* 161:355-363.
- Austerlitz, F., S. Mariette, N. Machon, P.-H. Gouyon and B. Godelle. 2000. Effects of colonization processes on genetic diversity: differences between annual plants and tree species. *Genetics* 154:1309-1321.
- Bacilieri, R., A. Ducousso, R.J. Petit and A. Kremer. 1996. Mating system and asymmetric hybridization in a mixed stand of European oaks. *Evolution* 50:900-908.
- Belward, A.S. (ed.). 1996. The IGBP-DIS global 1 km land cover data set (DISCover): Proposal and implementation plans. IGBP-DIS (International Geosphere-Biosphere Programme-Data and Information System) Working Paper No. 13, Toulouse, France.
- Berg, E. and F. Kuhlmann. 1993. Systemanalyse und Simulation für Agrarwissenschaftler und Biologen. Eugen Ulmer Verlag, Stuttgart, Germany.
- Berninger, F. 1997. Effects of drought and phenology on GPP in *Pinus sylvestris*: A simulation study along a geographical gradient. *Funct. Ecol.* 11:33-42.
- Beuker, E. 1994. Long-term effects of temperature on the wood production of *Pinus sylvestris* L. and *Picea abies* (L.) Karst. in old provenance experiments. *Scand. J. For. Res.* 9:34-45.
- Bossel, H. 1996. TREEDYN3 forest simulation model. *Ecol. Modell.* 90:187-227.
- Burczyk, J., W.T. Adams and J.Y. Shimizu. 1996. Mating patterns and pollen dispersal in a natural knobcone pine (*Pinus attenuata* Lemmon.) stand. *Heredity* 77:251-260.
- Bush, R.M., P.E. Smouse, W.T. Adams, S.H. Strauss, D.L. Copes and A.R. Griffin. 1992. Evidence for the adaptive significance of allozymes in forest trees. *New For.* 6:179-196.
- Caron, H., S. Dumas, G. Marque, C. Messier, E. Bandou, R.J. Petit and A. Kremer. 2000. Spatial and temporal distribution of chloroplast DNA polymorphism in a tropical tree species. *Mol. Ecol.* 9:1089-1098.
- Chen, C.W., W.T. Tsai and A.A. Lucier. 1998. A model of air-tree-soil system for ozone impact analysis. *Ecol. Modell.* 111:207-222.
- Chuine, I., J. Belmonte and A. Mignot. 2000. A modelling analysis of the genetic variation of phenology between tree populations. *J. Ecol.* 88:561-570.
- Collevatti, R.G., D. Grattapaglia and J.D. Hay. 2001. High-resolution microsatellite-based analysis of the mating system allows the detection of significant biparental inbreeding in *Caryocar brasiliense*, an endangered tropical tree species. *Heredity* 86:60-67.
- Darlington, C.D. 1939. The Evolution of Genetic Systems. Cambridge University Press, Cambridge, UK.
- Degen, B. and D.W. Roubik. 2004. Effects of animal pollination on pollen dispersal, selfing, and effective population size of tropical trees: a simulation study. *Biotropica* 36:165-179.
- Degen, B. and F. Scholz. 1996. Der Einsatz des Simulationsmodells ÖKO-GEN zur Erarbeitung von Entscheidungshilfen für eine nachhaltige Forstwirtschaft. Pp. 284-299 in Biodiversität und nachhaltige Forstwirtschaft (G. Müller-Starck ed.). Ecomed – Verlagsgesellschaft, Landsberg, Germany.
- Degen, B., E. Bandou and H. Caron. 2004. Limited pollen dispersal and biparental inbreeding in *Symphonia globulifera* in French Guiana. *Heredity* 93: 585-591.
- Degen, B., H.-R. Gregorius and F. Scholz. 1996. ECO-GENE, a model for simulation studies on the spatial and temporal dynamics of genetic structures of tree populations. *Silvae Genet.* 45:323-329.
- Degen, B., L. Llamas-Gomez and F. Scholz. 1999a. Erarbeitung von Entscheidungshilfen für eine nachhaltige Forstwirtschaft zum Schutze der genetischen Vielfalt von Waldbaum- und Waldstraucharten. Pp. 1-139 in Wichtige Einflussfaktoren auf die Biodiversität in Wäldern (F. Scholz and B. Degen eds.). Bundesforschungsanstalt für Forst- und Holzwirtschaft, Hamburg, Germany.
- Degen, B., D.W. Roubik and M.D. Loveless. 2002. Impact of selective logging and forest fragmentation on the seed cohorts of an insect-pollinated tree: a simulation study.

- Pp. 108-119 in *Modelling and Experimental Research on Genetic Processes in Tropical and Temperate Forests* (B. Degen, M.D. Loveless and A. Kremer eds.). EMBRAPA (A Empresa Brasileira de Pesquisa Agropecuária) Amazonia Oriental, Belém, Brazil.
- Degen, B., R. Streiff and B. Ziegenhagen. 1999b. Comparative study of genetic variation and differentiation of two pedunculate oak (*Quercus robur*) stands using microsatellite and allozyme loci. *Heredity* 83:597-603.
- Degen, B., R. Streiff, F. Scholz and A. Kremer. 1997. Analyzing the effects of regeneration regimes on genetic diversity and inbreeding in oak populations by use of the simulation model ECO-GENE. Pp. 9-21 in *Diversity and Adaption in Oak Species* (K.C. Steiner ed.). Pennstate, College of Agricultural Sciences, Pennsylvania, University Park, Pennsylvania, USA.
- Dick, C.W., G. Etchelecu and F. Austerlitz. 2003. Pollen dispersal of tropical trees (*Dinizia excelsa*: Fabaceae) by native insects and African honeybees in pristine and fragmented Amazonian rainforest. *Mol. Ecol.* 12:753-764.
- Doligez, A. and H.I. Joly. 1997. Mating system of *Carapa procera* (Meliaceae) in the French Guiana tropical forest. *Am. J. Bot.* 84:461-470.
- Doligez, A., C. Baril and H.I. Joly. 1998. Fine-scale spatial genetic structure with non-uniform distribution of individuals. *Genetics* 148:905-920.
- Everitt, J. and F. Judd. 1989. Using remote sensing techniques to distinguish and monitor black mangrove (*Avicennia germinans*). *J. Coast. Res.* 5:737-745.
- Fiorella, M. and W. Ripple. 1993. Determining successional stages of temperate coniferous forests with Landsat satellite data. *Photogramm. Eng. Rem. Sens.* 59:239-246.
- Foody, G. and R. Hill. 1996. Classification of tropical forest classes from Landsat TM data. *I. J. Rem. Sens.* 17:2353-2367.
- Forest Resource Assessment (FRA). 2000. Global Land Cover Characteristics Data Base Version 2.0, <http://edcdaac.usgs.gov/glcc/glcc.asp>.
- Friedman, S.T. and W.T. Adams. 1985. Levels of outcrossing in two loblolly pine seed orchards. *Silvae Genet.* 34:157-162.
- Geburek, T. and M. Mengel. 1998. Considerations in view of identity control of forest reproductive material and minimum number of seed trees required by law as exemplarily described in European larch. *Centralblatt für das Gesamte Forstwesen* 115:211-228.
- Giannini, R. and F. Magnani. 1994. Impact of global change on pollination processes and on the genetic diversity of forest tree populations. *For. Genet.* 1:97-104.
- Gill, D.E., L. Chao, S.L. Perkins and J.B. Wolf. 1995. Genetic mosaicism in plants and clonal animals. *Annu. Rev. Ecol. Syst.* 26:423-444.
- Glaubitz, J.C., J.C. Murrell and G.F. Moran. 2003. Effects of native forest regeneration practices on genetic diversity in *Eucalyptus consideniana*. *Theor. Appl. Genet.* 107:422-431.
- Gömöry, D. 1995. Simulation of the genetic structure and reproduction in plant populations: short note. *For. Genet.* 2:59-63.
- Gregorius, H.-R. 1978. The concept of genetic diversity and its formal relationship to heterozygosity and genetic distance. *Math. Biosci.* 41:253-271.
- Gregorius, H.-R. 1980. The probability of losing an allele when diploid genotypes are sampled. *Biometrics* 36:643-652.
- Gregorius, H.-R. 1991. Gene conservation and the preservation of adaptability. Pp. 31-47 in *Species Conservation: A Population-biological Approach* (A. Seitz and V. Loeschcke, eds.). Birkhauser Verlag, Basel, Switzerland.
- Gregorius, H.-R. and F. Bergmann. 1995. Analysis of isoenzyme genetic profiles observed in forest tree populations. Pp. 79-96 in *Population Genetics and Genetic Conservation of Forest Trees* (G. Muller Starck, ed.). SPB Academic Publishing, Amsterdam, The Netherlands.
- Grote, R. and M. Erhard. 1999. Simulation of tree and stand development under different environmental conditions with a physiologically based model. *For. Ecol. Manage.* 120:59-76.
- Hamrick, J.L. and M.J.W. Godt. 1990. Allozyme diversity in plant species. Pp. 43-63 in *Plant Population Genetics, Breeding, and Genetic Resources* (A.H.D. Brown, M.T. Clegg, A.L. Kahler and B.S. Weir, eds.). Sinauer Associates, Sunderland, USA.

- Hardy, O.J. and X. Vekemans. 1999. Isolation by distance in a continuous population: reconciliation between spatial autocorrelation analysis and population genetics models. *Heredity* 83:145-154.
- Helmer, E., S. Brown and W. Cohen. 2000. Mapping montane tropical forest successional stage and land use with multitemporal Landsat imagery. *J. J. Rem. Sens.* 21:2163-2183.
- Hosius, B. 1993. Wird die genetische Struktur eines Fichtenbestandes von Durchforstungseingriffen beeinflusst? *Forst und Holz* 48:306-308.
- Jaramillo-Correa, J.P., J. Beaulieu and J. Bousquet. 2001. Contrasting evolutionary forces driving population structure at expressed sequence tag polymorphisms, allozymes and quantitative traits in white spruce. *Mol. Ecol.* 10:2729-2740.
- Johnsen, K., L. Samuelson, R. Teskey, S. McNulty and T. Fox. 2001. Process models as tools in forestry research and management. *For. Sci.* 47: 2-8;
- Kanashiro, M., I.S. Thompson, J.A.G. Yared, M.D. Loveless, P. Coventry, R.C.V. Martins-da-Silva, B. Degen and W. Amaral. 2002. Improving conservation values of managed forests: the Dendrogene Project in the Brazilian Amazon. *Unasylva* 53:25-33.
- Kareinen, T., A. Nissinen and H. Ilvesniemi. 1998. Analysis of forest soil chemistry and hydrology with a dynamic model ACIDIC. *Acta For. Fenn* 262:42.
- Key, T., T.A. Warner, J.B. McGraw and M.A. Fajvan. 2001. A comparison of multispectral and multi-temporal information in high spatial resolution imagery for classification of individual tree species in a temperate hardwood forest. *Rem. Sens. Environ.* 75:100-112.
- Kramer, K. 1994. A modelling analysis of the effects of climatic warming on the probability of spring frost damage to tree species in The Netherlands and Germany. *Plant Cell Environ.* 17:367-377.
- Kremer, A. 2000. Global change and genetic diversity. *Revue Forestiere Francaise* 52:91-98.
- Kremer, A., J. Kleinschmit, J. Cottrell, E.P. Cundall, J.D. Deans, A. Ducousso, A. Konig, A. Lowe, J. Munro, R. Petit and B.R. Stephan. 2002. Is there a correlation between chloroplastic and nuclear divergence, or what are the roles of history and selection on genetic diversity in European oaks? *For. Ecol. Manage.* 156:75-87.
- Krusche, D. and T. Geburek. 1991. Conservation of forest gene resources as related to sample size. *For. Ecol. Manage.* 40:145-150.
- Landsberg, J.J. and R.H. Waring. 1997. A generalised model of forest productivity using simplified concepts of radiation-use efficiency, carbon balance and partitioning. *For. Ecol. Manage.* 95:209-228.
- Le Corre, V. and A. Kremer. 2003. Genetic variability at neutral markers, quantitative trait loci and trait in a subdivided population under selection. *Genetics* 164:1205-1219.
- Le Corre, V., N. Machon, R.J. Petit and A. Kremer. 1997. Colonization with long-distance seed dispersal and genetic structure of maternally inherited genes in forest trees: a simulation study. *Genet. Res.* 69:117-125.
- Ledig, F.T. 1986. Heterozygosity, heterosis, and fitness in outcrossing plants. Pp. 77-104 in *Conservation Biology: The Science of Scarcity and Diversity* (M.E. Soule, ed.). Sinauer Associates, Sunderland, USA.
- Liepert, S., R. Bialozyt and B. Ziegenhagen. 2002. Wind-dispersed pollen mediates postglacial gene flow among refugia. *Proc. Nat. Acad. Sci. USA* 99:14590-14594.
- Liesebach, M. 2002. On the adaptability of Norway spruce (*Picea abies* (L.) Karst.) to the projected change of climate in Germany. *Forstwissenschaftliches Centralblatt* 121:130-144.
- Liesebach, M., B. Degen and F. Scholz. 1999. Zur genetischen Anpassungsfähigkeit der Rotbuche. *Berichte über Landwirtschaft* 77:128-133.
- Lindner, M., B. Sohngen, L.A. Joyce, D.T. Price, P.Y. Bernier and T. Karjalainen. 2002. Integrated forestry assessments for climate change impacts. *For. Ecol. Manage.* 162:117-136.
- Matyas, C. 1994. Modelling climate change effects with provenance test data. *Tree Physiol.* 14:797-804.
- Michalek, J., N. French, E. Kasischke, R. Johnson and J. Colwell. 2000. Using Landsat TM data to estimate carbon release from burned biomass in an Alaskan spruce forest complex. *J. J. Rem. Sens.* 21:323-338.

- Murawski, D.A. and J.L. Hamrick. 1991. The effect of the density of flowering individuals on the mating systems of nine tropical tree species. *Heredity* 67:167-174.
- Nathan, R., U.N. Safriel and I. Noy-Meir. 2001. Field validation and sensitivity analysis of a mechanistic model for tree seed dispersal by wind. *Ecology* 82:374-388.
- Neale, D.B., M.M. Sewell, G.R. Brown, C. Plomion and C. Cahalan. 2002. Molecular dissection of the quantitative inheritance of wood property traits in loblolly pine. *An. For. Sci.* 59:595-605.
- Nikolov, N.T. and D.G. Fox. 1994. A coupled carbon-water-energy-vegetation model to assess responses of temperate forest ecosystems to changes in climate and atmospheric CO₂. Part I. Model concept. *Environ. Pollut.* 83:251-262.
- Nybom, H. 2004. Comparison of different nuclear DNA markers for estimating intraspecific genetic diversity in plants. *Molecular Ecology* 13: 1143-1155.
- Pax-Lenney, M., C. Woodcock, S. Macomber, S. Gopal and C. Song. 2001. Forest mapping with a generalized classifier and Landsat TM data. *Rem. Sens. Environ.* 77:241-250.
- Petit, R.J., E. Pineau, B. Demesure, R. Bacilieri, A. Ducousso and A. Kremer. 1997. Chloroplast DNA footprints of postglacial recolonization by oaks. *Proc. Nat. Acad. Sci. USA* 94:9996-10001.
- Phillips, P.D., T.E. Brash, I. Yasman, P. Subagyo and P.R. van Gardingen. 2003. An individual-based spatially explicit tree growth model for forests in East Kalimantan (Indonesian Borneo). *Ecol. Modell.* 159:1-26.
- Phillips, P.D., I.S. Thompson, J.N.M. Silva, P.R. van Gardingen and B. Degen. 2004b. Scaling-up models of tree competition for tropical forest population genetics simulation. *Ecol. Modell.* 180:419-434.
- Phillips, P.D., C.P. de Azevedo, B. Degen, I.S. Thompson, J.N.M. Silva and P.R. van Gardingen. 2004a. An individual-based spatially explicit simulation model for strategic forest management planning in the eastern Amazon. *Ecol. Modell.* 173:335-354.
- Porté, A. and H.H. Bartelink. 2002. Modelling mixed forest growth: a review of models for forest management. *Ecol. Modell.* 150:141-188.
- Pouliot, D., D. King, F. Bell and D. Pitt. 2002. Automated tree crown detection and delineation in high-resolution digital camera imagery of coniferous forest regeneration. *Rem. Sens. Environ.* 82:322-334.
- Pretzsch, H. 2000. From yield tables to simulation models for pure and mixed stands. *J. For. Sci.* 46:97-113.
- Pretzsch, H., P. Biber, J. Dursky, M. Lindner and W. Cramer. 2002. The single tree-based stand simulator SILVA: Construction, application and evaluation. *For. Ecol. Manage.* 162:3-21.
- Rajora, O.P. 1999. Genetic biodiversity impacts of silvicultural practices and phenotypic selection in white spruce. *Theor. Appl. Genet.* 99:954-961.
- Ritland, K. and S. Jain. 1981. A model for the estimation of outcrossing rate and gene frequencies using an independent loci. *Heredity* 47:35-52.
- Rossi, P., G.G. Vendramin and R. Giannini. 1996. Estimation of mating system parameters in two Italian natural populations of *Fagus sylvatica*. *Can. J. For. Res.* 26:1187-1192.
- Scholz, F. and B. Degen. 1999. Wichtige Einflussfaktoren auf die Biodiversität von Wäldern. Mitteilungen der Bundesforschungsanstalt für Forst- und Holzwirtschaft, Hamburg, Germany.
- Scholz, F., H.-R. Gregorius and D. Rudin. 1989. Genetic effects of air pollutants in forest tree populations. Springer-Verlag, Berlin, Germany.
- Schwalm, C.R. and A.R. Ek. 2001. Climate change and site: relevant mechanisms and modelling techniques. *For. Ecol. Manage.* 150:241-257.
- Schwartz, M.W., C.A. Bringham, J.D. Hoeksema, K.G. Lyons, M.H. Mills and P.J. van Mantgem. 2000. Linking biodiversity to ecosystem function: implications for conservation ecology. *Oecologia* 122:297-305.
- Scotti, I., G.G. Vendramin, L.S. Matteotti, C. Scarponi, M. Sari-Gorla and G. Binelli. 2000. Postglacial recolonization routes for *Picea abies* K. in Italy as suggested by the analysis of sequence-characterized amplified region (SCAR) markers. *Mol. Ecol.* 9:699-708.

- Skole, D. and C. Tucker. 1993. Tropical deforestation and habitat fragmentation in the Amazon: Satellite data from 1978 to 1988. *Science* 260:1905-1910.
- Smouse, P.E., R.J. Dyer, R.D. Westfall and V.L. Sork. 2001. Two-generation analysis of pollen flow across a landscape. I Male gamete heterogeneity among females. *Evolution* 55:260-271.
- Takahashi, M., M. Mukouda and K. Kono. 2000. Differences in genetic structure between two Japanese beech (*Fagus crenata* Blume) stands. *Heredity* 84:103-115.
- Takenaka, A. 2001. Individual-based model of a forest with spatial structure and gene flow. Pp. 415-420 *in* Present and Future of Modelling Global Environment Change: Toward Integrated Modelling (T. Matsuno and H. Kida, eds.). Terrapub, Tokyo, Japan.
- Thomasson, J., C.W. Bennett, B. Jackson and M. Mailander. 1994. Differentiating bottomland tree species with multispectral videography. *Photogramm. Eng. Rem. Sens.* 60.
- Vos, P., R. Hogers, M. Bleeker, M. Reijans, T. van der Lee, M. Hornes, A. Frijters, J. Pot, J. Peleman, M. Kuiper and M. Zabeau. 1995. AFLP: a new technique for DNA fingerprinting. *Nucleic Acids Res* 23:4407-4414.
- de Wasseige, C. and P. Defourny. 2002. Retrieval of tropical forest structure characteristics from bi-directional reflectance of SPOT images. *Rem. Sens. Environ.* 83:362-375.
- White, G.M., D.H. Boshier and W. Powell. 2002. Increased pollen flow counteracts fragmentation in a tropical dry forest: an example from *Swietenia humilis* Zuccarini. *Proc. Nat. Acad. Sci. USA* 99:2038-2042.
- Wolter, P., D. Mladenoff, G. Host and T. Crow. 1995. Improved forest classification in the northern lake states using multi-temporal Landsat imagery. *Photogramm. Eng. Rem. Sens.* 61:1129-1143.
- Woodcock, C., S. Macomber, M. Pax-Lenney and W. Cohen. 2001. Monitoring large areas for forest change using Landsat: generalization across space, time and Landsat sensors. *Rem. Sens. Environ.* 78:194-203.



Part 2

Case Studies from IPGRI's Research Project

**Introduction to IPGRI's role and
modus operandi, with special
reference to the BMZ project**

***Araucaria araucana* forest genetic
resources in Argentina**

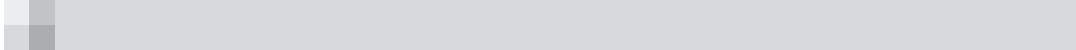
**Conservation, management and
sustainable use of *Araucaria
angustifolia* genetic resources
in Brazil**

**Genetic and ecological aspects of
nonwood forest product exploitation
in two western Amazonian
settlements**

**Conservation of Mata Atlântica
forest fragments in the State of
São Paulo, Brazil**

**A modelling case study: options
for FGR management in *Araucaria
araucana* ecosystems**

**Environmental heterogeneity shapes
genetic diversity through gene
flow in *Araucaria araucana* forest
ecosystems in Argentina**



Chapter 5

Introduction to IPGRI's role and *modus operandi*, with special reference to the BMZ project

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1. Introduction

This chapter focuses on forest genetic resources (FGRs) and associated conservation issues in South America. Special reference is made to the project funded by the German Federal Ministry for Economic Cooperation and Development (BMZ) in the conservation, management and sustainable use of FGRs in Brazil and Argentina, that was recently undertaken under the auspices of the International Plant Genetic Resources Institute (IPGRI).

2. The need for research on forest genetic resources

Forests fulfil basic life-support functions that include watershed protection, soil conservation, climate regulation, carbon sequestration, biodiversity conservation, and pest control (Salati *et al.* 1999). They also provide local communities and national economies with a variety of goods that include food, fodder, gum, fibre, fuelwood, timber and medicine. As a consequence, they play primary roles in both subsistence and modern socioeconomic development in many parts of the world. Forests also contribute to the cultural well-being of countless societies.

Owing to their interdependence within the hierarchy of biological organization, ecosystems, species and genes represent the principal targets for biodiversity conservation. Ultimately, the capacity of plants to adapt to changing environmental conditions and continue evolving is dependent upon their infraspecific genetic variation. Genetic variation also constitutes the foundation of breeding programmes that aim to meet society's ever-changing needs. FGRs, which are defined here as 'the present and potential value of the heritable materials contained within and among forest species', are among society's most valuable sustainable assets. However, in the past few decades, the threat to FGRs in the tropics has been growing dramatically. Severe and continuous tropical forest depletion is now occurring around the world. This results from unsustainable timber extraction, other types of forest habitat degradation, and conversion of forests to other land uses.

These changes have directly affected the amounts and patterns of genetic diversity in forest trees. Many populations and species possessing unique and useful properties are being lost, sometimes without even having been identified and studied by science (Mittermeier *et al.* 1999). While degraded ecosystems can sometimes be rehabilitated – often with difficulty and at great cost – extinction of species or the loss of unique genetic diversity within species is permanent. This can potentially have significant effects on future uses and development. As a result, the conservation and sustainable use of FGRs has become a pressing and vital requirement for all local, regional, national and international forest stakeholders (Ouédraogo 1997; Convention on Biological Diversity 2004). The task at hand is complicated by the fact that conservation strategies cannot aim to conserve the

genetic makeup of forest species as they exist today – adapted to current environmental conditions, but rather they must be designed to maintain the evolutionary processes that will enable species to adapt to future conditions (Namkoong 1986; Eriksson *et al.* 1993). A central issue in FGR conservation is thus the definition of the amount and the pattern of genetic variation that should be maintained for forest species to meet this requirement (Palmberg 1999). Equally important is the work, whenever possible, of gene conservation programmes to prevent or reduce species extinctions, genetic erosion, and strong directional change in the genetic composition of conserved populations (FAO 1993).

3. Conserving FGRs *in situ*

Forest scientists and the wider conservation and research communities are now involved in ongoing efforts to develop conservation and sustainable use strategies for biodiversity. Conservation can either be achieved through *in situ* or *ex situ* approaches (Ledig 1986), but the two should be implemented in a complementary manner whenever this is needed and possible. Nevertheless, the preferred strategy is often *in situ* conservation of FGRs, that is, conservation of FGRs within their original/natural environments, as this has several advantages (Frankel 1976). This is because *in situ* conservation of FGRs allows for the maintenance of basic evolutionary processes and it provides breeders with a dynamic source of variation. *In situ* conservation is especially recommended for species that cannot be established outside of their natural habitats, that require complex ecosystem interactions to subsist, and for those that produce recalcitrant seeds. Moreover, *in situ* conservation programmes designed specifically for forest species can often result in the protection of many other plant and animal species and even the ecosystems in which these species occur. Lastly, if practised correctly, *in situ* conservation allows for FGRs to be used while they are being sustained. Therefore, during conservation planning, the sustainable use of resources by stakeholders should be factored into decisions, thereby also potentially improving the cost-effectiveness and the political support for conservation.

It is worth emphasizing that the conservation and sustainable use of FGRs is not just a problem for the future. It affects millions of people now. Because of this, conservation cannot be viewed separately from development, that is, from human needs and demands, and this is perhaps the greatest challenge to *in situ* FGR conservation. Reconciling conservation activities with immediate human needs, recognizing the importance of different stakeholders for their roles in maintaining and managing genetic diversity, and reducing tensions between biological and socioeconomic objectives are and will be areas in which forestry managers and researchers must invest themselves.

Production of forest-derived goods and environmental services will require that basic genetic and silvicultural guidelines be incorporated into forest management prescriptions and interventions (Kemp 1992). Conservation of genetic resources through sustainable management is important, particularly for the large majority of forest tree species that are not being grown in plantation or domestication programmes (National Research Council, Board of Agriculture 1991).

Nevertheless, while *ex situ* conservation methods are fairly well developed from a technical point of view and are widely applied around the world, relatively little *in situ* FGR conservation has occurred, especially as it applies to conservation of infraspecific diversity. In part, this may be due to the prevailing attention given to *ex situ* approaches, or to the still limited cooperation among breeders, foresters and conservationists. Moreover, the earliest efforts in *in situ* conservation were oriented towards the protection of endangered species and intact ecosystems, with much less emphasis placed on within-species diversity (FAO 1989). Nevertheless, during the last 20 years or so, awareness has grown that effective *in situ* conservation, management and sustainable use of plant and animal species must employ strategies that include information on the dynamics of infraspecific variation. In spite of such general progress in the philosophy of conservation and sustainable use, advances in *in situ* theory and method that can be applied to on-the-ground FGR programmes have been slow in coming. This is mostly due to limited knowledge about the extent of genetic diversity in forest trees, its spatial and temporal

organization, the genetic processes in trees, and the impact of human activities on FGRs. In this last regard, investigation of the role of local communities in the selective regeneration, dissemination and planting of forest germplasm is a high priority.

4. Challenges for conserving FGRs in the tropics

Genetic resources refer to all the genetic variation contained within and among forest species. In practice, a very large number of woody and herbaceous species which constitute important economic resources are covered by the term 'forest genetic resources'. While research has shown that tropical forest tree species possess high levels of genetic diversity, maintain large amounts of genetic variation within and among populations, and are predominantly outcrossed and have high levels of gene flow (Janzen 1971; Bawa *et al.* 1985), for FGR conservation purposes, two further areas of research are urgently needed. Much more needs to be known about the relationship between human activities and forest biodiversity and, in the same context, about the extent, distribution and structure of infraspecific genetic diversity within trees subject to human actions.

Those forests referred to as lowland tropical rain forests contain the greatest number of species, and their destruction would have the most serious global effects (Whitmore and Sayer 1992; Alvarez-Buylla *et al.* 1996). Conservation of tropical forests is thus one of the most urgent and challenging tasks of foresters, biologists, social scientists, environmental managers, entrepreneurs, nonprofit organizations and public administrators. The challenging aspects are the high species numbers, their level of threat, the presence of many noncommercial species, the difficulties in regenerating target species, soil fragility and access issues.

In situ conservation of FGRs has been slower in the tropics in comparison to temperate areas because of the following limitations and challenges (Bawa 1997; Ouédraogo 1997):

- The very large number of species that occur in tropical forests. This has created problems in priority-setting for both research and conservation and, as a consequence, in obtaining funds for this work;
- The often modest infrastructures of local institutions. The lack of trained research and conservation personnel and the high costs of training these people have contributed to the slow progress in generating knowledge about the taxonomy, biology and genetics of tropical forest species, and in developing effective management programmes;
- The diverse biological characteristics of tropical forest trees and their complex ecological relationships. Features of tropical forest trees, including sophisticated pollinator requirements and interactions, highly variable densities and spatial distributions of species, mixed mating systems, and diverse phenological patterns often require that conservation methods be tailored to individual species;
- The long-lived nature of trees can produce an illusion of forest health. Because tropical forests can persist for long periods of time without regenerating themselves, an impression of persistence can be conveyed when, in reality, the population is being genetically eroded or is even moving towards extinction (the 'living dead' of Janzen 1986);
- Evaluation and comparison of morphological and heritable characteristics have only been undertaken for a few species. The tools and technologies to assess genetic diversity in tropical trees are recent and still developing. Genetic markers are limited;
- Limited income options for forest-dependent communities. Limited employment or income options have led many forest-dwelling communities to adopt short-term forest management perspectives or even abusive practices. The development of effective forest management partnerships among local communities, indigenous people, governments and other stakeholders is an ongoing challenge that often requires specialized skills;
- National policies and legal frameworks conducive to conservation and sustainable use of FGRs are often lacking. Complications generated by variable interests and actors, and differences among traditional, local and national laws have often been difficult to overcome.

Many of the tree species found in rapidly disappearing tropical forest habitats require immediate conservation measures. These must be based on an understanding of the factors that will permit the long-term *in situ* conservation of genetic diversity in forest tree populations. And knowledge needs to be generated about the kinds and levels of genetic variation that are required to capture different types of adaptability. At the ecosystem level, knowledge of the processes that affect genetic diversity and particularly the genetic effects of human activities will be fundamentally important to the development of complementary conservation strategies. This same knowledge is needed to develop effective policies that can address the different needs and interests of local communities and governments (Cossalter *et al.* 1993; FAO 1993; Kageyama 1997; Palmberg-Lerche 1997).

In order for it to work, conservation of FGRs must be integrated into an overall framework of sustainable forest management. In the past, efforts to conserve protected areas such as national parks or reserves did not give sufficient attention to the distribution and infraspecific variation of species. Information from both of these areas is central to ensuring the maintenance of the adaptive capacity and the productive potential of trees to meet present and future needs (Kemp 1992; FAO 1993; McNeely 1993; Bawa 1997; BMZ 1997).

On the technical side, studies that demonstrate the compatibility of genetic conservation with managed use of forest resources need urgent implementation at national, regional and local levels, with the results disseminated widely. On the management side, integrated strategies for the conservation and use of FGRs must also be developed. Thus, priority should be given both to the research needed to identify patterns of genetic variation as well as underlying evolutionary changes in forest ecosystems, and to the development of practical conservation procedures. Enough flexibility must be built into the conservation strategies to permit management strategies to change as new findings become available (Ouédraogo 1997). In addition, concerted actions are needed to strengthen local, regional and national forest conservation capacities through information and technology transfers, and through networking and collaboration among countries and nongovernmental organizations (NGOs). IPGRI is actively involved in tropical forest research and in related capacity building around the world. A more detailed description of some of IPGRI's research projects and their practical applications is presented in Chapter 12.

5. IPGRI's activities in FGR research and conservation

IPGRI is part of the Consultative Group on International Agricultural Research (CGIAR), a strategic alliance of countries, international and regional organizations, and private foundations supporting 15 international agricultural centres. These centres work with national agricultural research institutions and agencies, civil society organizations and the private sector to mobilize agricultural science to reduce poverty, foster human well-being, promote agricultural growth and protect the environment. Box 1 describes the way in which IPGRI works.

Research at IPGRI includes a programme on FGRs that has several priorities:

- to study patterns of and threats to genetic diversity in forest tree species;
- to investigate biological processes that regulate genetic diversity;
- to examine the effects of human activities on genetic processes;
- to document local resource use and to analyze the consequences of FGR losses on rural livelihoods.

The IPGRI FGR programme also:

- establishes guidelines for FGR conservation, restoration and sustainable use;
- analyzes policy and socioeconomic issues related to FGR conservation and use;
- devises participatory FGR conservation schemes;
- creates or selects methods that help IPGRI determine priority species and populations for research and conservation.

Box 1. IPGRI's mode of operation

IPGRI works through partnerships, complementing its partners' resources to provide a collaborative research and conservation effort that is highly cost-effective. Savings accrue through the sharing of such things as staff time, laboratory and field facilities, and information technology. In addition, technology and information are transferred to partners who have limited resources. IPGRI operates directly or as a catalyst within almost all areas of the research and development processes related to the conservation and use of plant genetic resources (PGRs). With its partners, IPGRI identifies those places within recognized research and conservation priorities where its involvement would bring maximum gain. In this way, IPGRI and its partners optimize returns from resources invested in what are truly collaborative efforts. In undertaking its research and conservation programmes, IPGRI particularly values operating within networks, as networks help ensure that work occurs cost-effectively and that materials, methods, results and information are disseminated widely. IPGRI also emphasizes research that is undertaken with partners, as this helps ensure that projects are well designed, and that local facilities, human resources and a willingness to undertake research activities are available for its implementation. Nevertheless, constraints and challenges are always encountered in PGR research and *in situ* PGR conservation. Several examples of FGR research and *in situ* conservation are illustrated in this book.

Because *in situ* conservation of FGRs involves genetics, ecology and human social systems, it requires a multidisciplinary approach that integrates biological and social science disciplines and strategies. Technical and scientific knowledge, regional collaboration, appropriate policies, public awareness, and commitment at local, regional, national and international levels are needed to create effective *in situ* conservation programmes that simultaneously promote sustainable livelihoods.

5.1 Background and design of IPGRI's BMZ-funded research project on FGRs in South America

The Convention on Biological Diversity provides a framework for the protection of forest ecosystems and their tree species (see the expanded programme of work on forest biological diversity adopted in 2002 by the Conference of the Parties at its sixth meeting, in annex to decision VI/22). Countries that ratified the Convention are required to assess and monitor their biological resources and to develop effective strategies for their conservation. As a result, the conservation and sustainable use of forests have become priorities for many regional, national and international forestry programmes. But our limited knowledge of the impact of uncontrolled exploitation, deforestation and other threats to forest diversity seriously limits the capacity of agencies and institutions to plan and execute appropriate actions. The development of strategies for the conservation and sustainable use of forests should ensure a fair distribution of their resources to the livelihoods of local communities and to national economies while securing the genetic diversity of target species for the future. This formula is particularly important in regions and countries where rural and urban populations are heavily dependent on forests (CIFOR/IPGRI 1997).

IPGRI has carried out research in South America since the early stages of its FGR programme, established in 1993. Since then, work has steadily increased in the region, most of it in synergy with the IPGRI regional office in Cali, Colombia. This office has a high-level research capability on site, and research focuses mainly on the effects of anthropogenic activities on forest genetic diversity.

The BMZ-funded project on FGRs was conceived as a response to a demand for appropriate strategies for FGR conservation and sustainable use in South America

and, with its partners, IPGRI planned and implemented a range of research activities in forest ecosystems in the region. The objectives of the project were to generate scientific knowledge and develop practical procedures that could improve understanding of sustainable forest management, assist forestry professionals at all levels, and enhance the well-being of the communities of users (IPGRI 1998). By improving our understanding of the impact of human activities on the genetic diversity and ecological processes in selected forest ecosystems, guidelines and tools could be developed to assist decision-makers and forest managers in both forest policy development and its implementation.

The research project was designed to promote collaborative efforts among national programmes in Brazil and Argentina, German research institutes and universities, and CGIAR centres. It built on existing collaborations among these countries and institutions and established new partnerships, especially with local communities. A multidisciplinary approach was adopted.

The partnership consisted of an interdisciplinary research team of social scientists, ecologists, entomologists and geneticists. The project sought the active participation of local groups of forest users in Brazil and Argentina, where both rural and urban populations are heavily dependent on rapidly diminishing forests for a multitude of goods and services. Box 2 gives more information about the project's design. The formal and informal collaborating institutions and organizations that took part in the project are listed in Table 1. The principal scientists and their general areas of research are summarized in Table 2 (on p. 100). Figure 1 is a photo of some of the project participants.

The project was divided into three thematic components: one that explored socioeconomic issues and policy, one that studied the ecology of 'model' species with variable characteristics in forest ecosystems near settlements with different socioeconomic conditions, and one that explored genetic features of the selected species. Agreement on research methods (e.g., sample sizes, genetic parameters, structure of socioeconomic surveys, etc.) was reached through preliminary planning meetings. Key partners in the project agreed on the research hypotheses and on the target ecosystems and species, as well as on the project components with highest priority.

Since the project structure was particularly complex, involving different types of expertise and research methods, the ZOPP project planning method was used (ZOPP – Zielorientierte Projektplanung – in German, or GOPP – Goal Oriented Project Planning – as it is sometimes presented in English; for more information see the website:

Box 2. Principles underlying the design of the BMZ-funded FGR project

- Participatory approaches are to be used by national and international stakeholders, that is, forest-dependent communities must be involved in project development, implementation and outcome
- A multidisciplinary perspective should be adopted, linking conservation and use of FGRs within socioeconomic contexts
- Diverse (even contrasting) ecosystems are to be selected for study (e.g., from simple to complex; disturbed to less-disturbed to intact) for comparative purposes
- Research is to be undertaken at the local level, and designed to address FGR conservation and use problems of global significance
- Species with contrasting life histories and reproductive strategies are to be targeted, for comparative purposes
- Studies should be devised to generate knowledge about patterns of genetic variation, and on processes that affect genetic diversity
- Flexible methods and strategies, which could be modified as needed, should be developed
- Practical tools and procedures in conservation and sustainable forest management are to be developed with an objective of wide application

Table 1. Collaborating institutions and organizations that participated in the BMZ-funded research project.

Type of organization or institution	Country	Names of organizations and institutions
International NGO		<ul style="list-style-type: none"> • International Plant Genetic Resources Institute (IPGRI), • Centre for International Forestry Research (CIFOR)
National Institution	Germany	<ul style="list-style-type: none"> • Federal Research Centre for Forestry and Forest Products (BFH) • Institute for Forest Genetics and Forest Tree Breeding • Institute for World Forestry
	Brazil	<ul style="list-style-type: none"> • Universidade Federal do Paraná, Departamento de Silvicultura e Manejo • Instituto Ambiental do Paraná • Universidade de São Paulo, Escola Superior de Agricultura Luis De Queiroz, Departamento de Ciências Florestais • Centro Nacional de Recursos Genéticos e Biotecnologia (CENARGEN), Brasília • Universidade Federal de Uberlândia
	Argentina	<ul style="list-style-type: none"> • Instituto Nacional de Tecnología Agropecuaria (INTA), Unidad de Genética Forestal, INTA-EEA Bariloche, San Carlos de Bariloche • Grupo Forestal, INTA-EEA Montecarlo, Misiones • Administración Parque Nacional, Delegación Técnica Regional Patagonia. Parque Nacional Lanín • Centro de Investigación y Extensión Forestal Andino-Patagónico (CIEFAP) • Universidad Nacional de Misiones. Facultad de Ciencias Forestales, El Dorado
Regional NGO		<ul style="list-style-type: none"> • Conselho Nacional dos Seringueiros, Acre, Rio Branco, Brazil • Instituto de Projetos e Pesquisas Ecologicas (IPE), Brazil • Movimento dos Trabalhadores Rurais Sem Terra (MST), Brazil



Figure 1. Project participants at one of the four project regional workshops held in June 2002, during a field trip (Iraty, Paraná State, Brazil; photo: B. Vinceti)

Table 2. BMZ-funded project framework, including countries or partners, project components with personnel involved and their expertise.

Countries or partners	Socioeconomic research (social anthropology, sociology, economics)	Ecological research (ecology, biology, silviculture- forest management)	Genetic research (population genetics conservation, reproductive biology)	Capacity building activities (training, education and information, institutional strengthening)
BRAZIL Tropical/Atlantic Tropical/Amazon Subtropical	D. Caron C. Udri O. Queda D. Blondi Batista C. Sonda	G. Flavio L. Gonzaga R. Seitz* A. Higa*	P.Y. Kageyama P.E. Oliveira** D. Grattapaglia R. Stephan* A. Higa**	Univ. São Paulo Univ. Fed. Uberlândia** Museum Goldi, Belém Univ. Fed. Paraná
ARGENTINA Subtropical	L. Colcombet G. Huerta	R. Fernández J. Sanguinetti	G. Rodriguez L. Gallo	INTA-Bariloche INTA-Misiones
GERMANY	P. Elsasser*	H. Ellenberg* J. Heuveldop*	R. Stephan* A. Koenig* B. Ziegenhagen**	BFH
IPGRI	P. Eyzaguirre J. Engels	A.S. Ouédraogo L. Petri* B. Vinceti**	A.S. Ouédraogo T. Hodgkin L. Guarino	A.S. Ouédraogo R. Lastra
CIFOR	J. Poulsen	J. Poulsen / C. Sebogal		

* = partner withdrew; ** = partner joined later.

<http://www.worldbank.org/wbi/sourcebook/sba102.htm>). ZOPP's output is a planning matrix – the logical project framework – which summarizes and structures the main elements of a project and highlights logical links between intended inputs, planned activities and expected results. This tool was adopted during the second planning meeting in order to facilitate the selection of sound objectives, appropriate species, activities to be carried out and research techniques to be used to achieve the objectives of the project (GTZ 1988).

The first (socioeconomic) component of the project addressed issues in policy and human resource development for conservation, and particularly focused on the following objectives: (i) document the relative contributions of timber and nonwood forest products to household incomes and to other off-farm and on-farm income sources; (ii) identify the economically important tree species and document the patterns of land-use intensity and change, especially including logging, livestock grazing, gathering nonwood forest products and rubber tapping; (iii) examine alterations in livelihoods resulting from recent social and economic changes and the consequences for FGR use; (iv) assess the impact of current land use and forestry policies on timber and nonwood forest product management; and (v) propose options favouring sustainable use.

The second (ecological) component of the project examined the conservation status of selected forest stands and tree species and investigated the demographic structure of selected populations of those species. Work was specifically directed towards determining the effect of human activities on the selected trees in terms of their reproductive biology and ecology, phenology, pollinator foraging behaviour, seed dispersal, seed bank development, seed and seedling predation, and other regeneration dynamics.

The third (genetic) component of the project analyzed patterns of genetic variation within the selected tree species and examined the effects of human activities like logging and gathering on the genetic diversity and genetic structure of those species, including nonwood forest product species.

The project included activities that emphasized knowledge transfer among research teams from different countries. It also sought to promote exchange of views and information among scientists, local policy-makers and forest-user groups, focusing particularly on strategies and methods for conservation and use of FGRs.

The working hypotheses to be tested were:

1. The types and intensities of forest resource use by local communities will depend on such socioeconomic factors as access to markets and availability of alternative employment (local or urban).
2. Derived from cultural background and economic conditions, the value attributed to the forest by local communities will determine the degree to which the forest is managed sustainably.
3. Increased levels of forest use will affect both the extent and the distribution of infraspecific diversity in tree species.
4. The impact of forest use on the genetic diversity of tree species will depend on the type and the intensity of forest use and the ecology of the species.
5. Broad predictive relationships among patterns of use, species ecology and changes in genetic diversity are identifiable.
6. A key biological factor that mediates the impact of use on the genetic diversity of the species is its reproductive ecology, including pollination and mating system and seedling recruitment.
7. Information on changing socioeconomic conditions and related human activities and their effects on genetic resources is important for generating sustainable management options for the future.

Three major forest ecosystems and clusters of tree species were selected to test these hypotheses (listed in Table 3, overleaf):

- Threatened *Araucaria* spp. forest ecosystems in southern Brazil and northeastern and southwestern Argentina (1% of original forest remaining)
- Endangered Atlantic forest ecosystems of Brazil (7% of original forest remaining)
- Highly diverse and relatively intact Brazilian Amazon tropical forest ecosystems, in areas with recently established and active sawmill industries and logging operations.

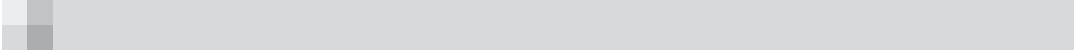


Table 3. Target forest ecosystems, locations and species chosen for BMZ-funded multidisciplinary research.

Ecosystems	Locations/sites	Model/target species
Araucaria forests	Northeastern and southwestern Argentina and southern Brazil	<i>Araucaria angustifolia</i> , <i>A. araucana</i> , <i>Podocarpus lambertu</i> , <i>Ocotea porosa</i> , <i>Cedrela fissilis</i> , <i>Ilex paraguariensis</i>
Atlantic forest	Rio Doce, Brazil ~30 000 ha	<i>Maytenus</i> spp., <i>Dalbergia nigra</i> , <i>Croton</i> spp., <i>Euterpe oleracea</i> , <i>Cedrela fissilis</i> , <i>Hymenaea courbaril</i> , <i>Copaifera langsdorfii</i>
Amazon forest	Acre, Brazil ~1 000 000 ha	<i>Euterpe oleracea</i> , <i>Visnia</i> spp., <i>Hevea brasiliensis</i> , <i>Cedrela odorata</i> , <i>Hymenaea courbaril</i> , <i>Copaifera langsdorfii</i>

The principal outcomes of the project are illustrated in the rest of Part 2 in this book. Chapters 6 and 7 focus on two types of araucaria forest ecosystems in Argentina and Brazil. The case study presented in Chapter 6 looks at the impact of current patterns of forest use by local indigenous communities on the conservation status of *Araucaria araucana* FGRs within and in areas immediately neighbouring a protected area in northern Patagonia (Argentina). Chapter 7 illustrates the findings of an assessment on the conservation status of another araucaria species, *A. angustifolia*, in the State of Paraná, Brazil. The research carried out within the *A. angustifolia* subproject mainly investigated the repercussions of different policy frameworks regulating access to and use of FGRs on the conservation status of *A. angustifolia*. In Chapter 8, different extraction regimes of nonwood forest products, derived from selected species within the Amazon forest ecosystem, are compared in the light of their influence on genetic processes. Chapter 9 presents an extreme case of forest degradation in the southeastern part of the State of São Paulo (Brazil) caused by heavy and rapid fragmentation, and analyzes the conditions of FGRs in priority species inside fragments of different size and with varied use histories.

Finally, in Chapters 10 and 11, two examples of modeling applied to *A. araucana* forest ecosystems are illustrated. The first provides a landscape-scale view of the dynamic processes affecting araucaria FGRs, describing links and feedback mechanisms between (i) the conservation status of araucaria FGRs inside a protected area; (ii) the socioeconomic dynamics of local indigenous communities that are partly dependent on nonwood forest products and on forest services; and (iii) various productive activities carried out by the local communities that affect araucaria genetic resources. Chapter 11 uses a modeling application to explain and predict patterns of araucaria spatial genetic structure, based on the analysis of selection pressures, gene flow and species distribution patterns.

References

Alvarez-Buylla, E.R., R. García-Barros, C. Lara-Moreno and M. Martínez-Ramos. 1996. Demographic and genetic models in conservation biology: applications and perspectives for tropical rainforest tree species. *Annu. Rev. Ecol. Syst.* 27:387-421.

Bawa, K.S. 1997. *In situ* conservation of tropical forests. Pp. 5-6, in Focus paper, IPGRI Forgen News, the International Plant Genetic Resources Institute (IPGRI), Rome, Italy.

Bawa, K.S., Perry, D.R. and Beach, J.H. 1985. Reproductive biology of tropical lowland rainforest trees 1. Sexual systems and incompatibility mechanisms. *Am. J. Bot.* 72:331-345.

- BMZ. 1997. Tropical forest conservation and German development cooperation, Federal Ministry for Economic Cooperation and Development (BMZ). Bonn, Germany.
- CIFOR/IPGRI (Centre for International Forestry Research/International Plant Genetic Resources Institute). 1997. Impact of disturbance on genetic resources of tropical forests. Research Update No. 3. Bogor, Indonesia/ Rome, Italy.
- Convention on Biological Diversity. 2004. Expanded programme of work on forest biological diversity. Montreal, Canada.
- Cossalter, C., L. Thomson, P. Tompsett and T. Vercoe. 1993. Forest genetic resources – an agenda for action. Consultants' synthesis report to CIFOR, IPGRI and ICRAF on Forestry/ Agroforestry Genetic Resources. International Plant Genetic Resources Institute (IPGRI), Rome, Italy.
- Eriksson, G., G. Namkoong and J.H. Roberds. 1993. Dynamic gene conservation for uncertain futures. *For. Ecol. Manage.* 62:15-37.
- FAO (Food and Agriculture Organization of the United Nations). 1989. Plant Genetic Resources: Their conservation *in situ* for human use. FAO, Rome, Italy.
- FAO (Food and Agriculture Organization of the United Nations). 1993. The challenge of sustainable forest management – what future for the world's forests? FAO, Rome, Italy.
- Frankel, O.H. 1976. Natural variation and its conservation. Pp. 21-44 *in* Genetic Diversity in Plants (M.R. Askel and I. von Bostel, eds.). Plenum Press, New York, USA.
- GTZ (Deutsche Gesellschaft für Technische Zusammenarbeit). 1988. ZOPP (An introduction to the method). GTZ, Eschborn, Germany.
- IPGRI (International Plant Genetic Resources Institute). 1998. Conservation, management and sustainable use of forest genetic resources with reference to Brazil and Argentina. A restricted Research Agenda project proposal submitted to the Bundesministerium für Wirtschaftliche Zusammenarbeit und Entwicklung by IPGRI, Rome, Italy.
- Janzen, D.H. 1971. The euglossine bees as long-distance pollinators of tropical plants. *Science* 171:203-5.
- Janzen, D.H. 1986. The future of tropical ecology. *Annu. Rev. Ecol. Syst.* 17:305-324.
- Kageyama, P.Y. 1997. Parameters for *in situ* conservation in tropical forests of Brazil. International Plant Genetic Resources Institute (IPGRI) project progress report. IPGRI, Rome, Italy.
- Kemp, R.H. 1992. The conservation of genetic resources in managed tropical forests. *Unasylva* 43(169):34-40.
- Ledig, F.T. 1986. Conservation strategies for forest gene resources. *For. Ecol. Manage.* 14:77-90.
- McNeely, G.A. 1993. Lessons from the past: forests and biodiversity. Proceedings of the Global Forest Conference, February 1993, Bandung, Indonesia.
- Mittermeier, R.A., N. Myers, P.R. Gil and C.G. Mittermeier. 1999. Hotspots: Earth's biologically richest and most endangered terrestrial ecoregions. CEMEX and Conservation International, Japan.
- Namkoong, G. 1986. Genetics and the forests of the future. *Unasylva* 38(152):2-18.
- National Research Council, Board of Agriculture. 1991. Managing global genetic resources: forest trees. Forest Genetic Resources workgroup. National Academy Press, Washington DC, USA.
- Ouédraogo, A.S. 1997. Conservation, management and use of forest genetic resources. Proceedings of XX World Forestry Congress, 13-22 October 1997, Antalya, Turkey. 2:173-188
- Palmberg, C. 1999. Conservation and management of forest genetic resources. *J. Trop. For. Res.* 11(1):286-302.
- Palmberg-Lerche, C. 1997. Conservation and sustainable utilization of forest genetic resources. Presentation notes at COFO (Committee on Forestry) meeting. 10-13 March 1997, Food and Agriculture Organization, Rome, Italy.
- Salati, E., W.A.N. Amaral and A. Santos. 1999. Investing in carbon storage: a review of Brazilian forest projects. Pp. 101-114 *in* Promoting Development while Limiting Greenhouse Emissions: Trends and Baselines (J. Goldemberg and W. Reid, eds.). United Nations Environment Programme (UNEP), New York, USA.

Whitmore, T.C. and J.A. Sayer. 1992. Deforestation and species extinction in tropical moist forests. *In* Tropical Deforestation and Species Extinction. IUCN Conservation Programme (T.C. Whitmore and J.A. Sayer, eds.). The World Conservation Union (IUCN) and Chapman and Hall, London, UK.

Chapter 6

***Araucaria araucana* forest genetic resources in Argentina**

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1. Introduction

This chapter investigates threats to the genetic resources of *Araucaria araucana*, studied as part of a project funded by the German Federal Ministry for Economic Cooperation and Development (BMZ) at a site in Argentina. Genetic processes were examined in *A. araucana* forests that were differently affected by human activities and which grew along an environmental gradient. These processes were then analyzed in relation to biological dynamics and socioeconomic conditions. Suggestions are made on how to incorporate research results into guidelines for sound management of *A. araucana* forest genetic resources (FGRs).

2. Description of the case study

Araucaria araucana (common name: araucaria, or pehuén in the Mapuche language) is a long-lived, canopy-tree conifer with a typical large umbrella-like form when mature. It is a member of the most ancient extant family of seed plants, Araucariaceae (Tomlinson 2002), and specimens as old as 1300 years have been recorded (Montaldo 1974). Trees can grow up to 50 m in height and up to 2.5 m in diameter (Veblen and Delmastro 1976). The species is endemic to the northern regions of the temperate sub-Antarctic forests of Chile and Argentina. Figure 1 (overleaf) shows its regional distribution in Argentina (northern Patagonia) and the location of the study sites. Araucaria is of high conservation concern because of its restricted distribution, slow growth and low regenerative capacity, making it particularly susceptible to human impact.

The species is also important for its high economic and cultural value to the Mapuche people, ancient inhabitants of southern Chile and Argentina (Box 1, on p. 107). Livestock grazing, fuelwood exploitation and araucaria seed collection are the main activities upon which the economy of the Mapuche tribes is based. The araucaria seed ('piñon' in Spanish) is an important resource for Mapuche communities where it is consumed by humans and used as a livestock feed, with surplus usually sold in markets. The Mapuche communities practise transhumance, or seasonal pastoralism involving movement of humans along with the pastured animals. For most of the year people live in the steppe, a dry environment composed mainly of tussock grasses and shrubs, where isolated stands of araucaria are found, but some families move with their livestock to summer grazing grounds high in the mountains near or in araucaria and *Nothofagus* spp. association forests. Grazing occurs there in conjunction with fuelwood gathering and seed collecting while people are in these areas, known locally as 'veranada'.

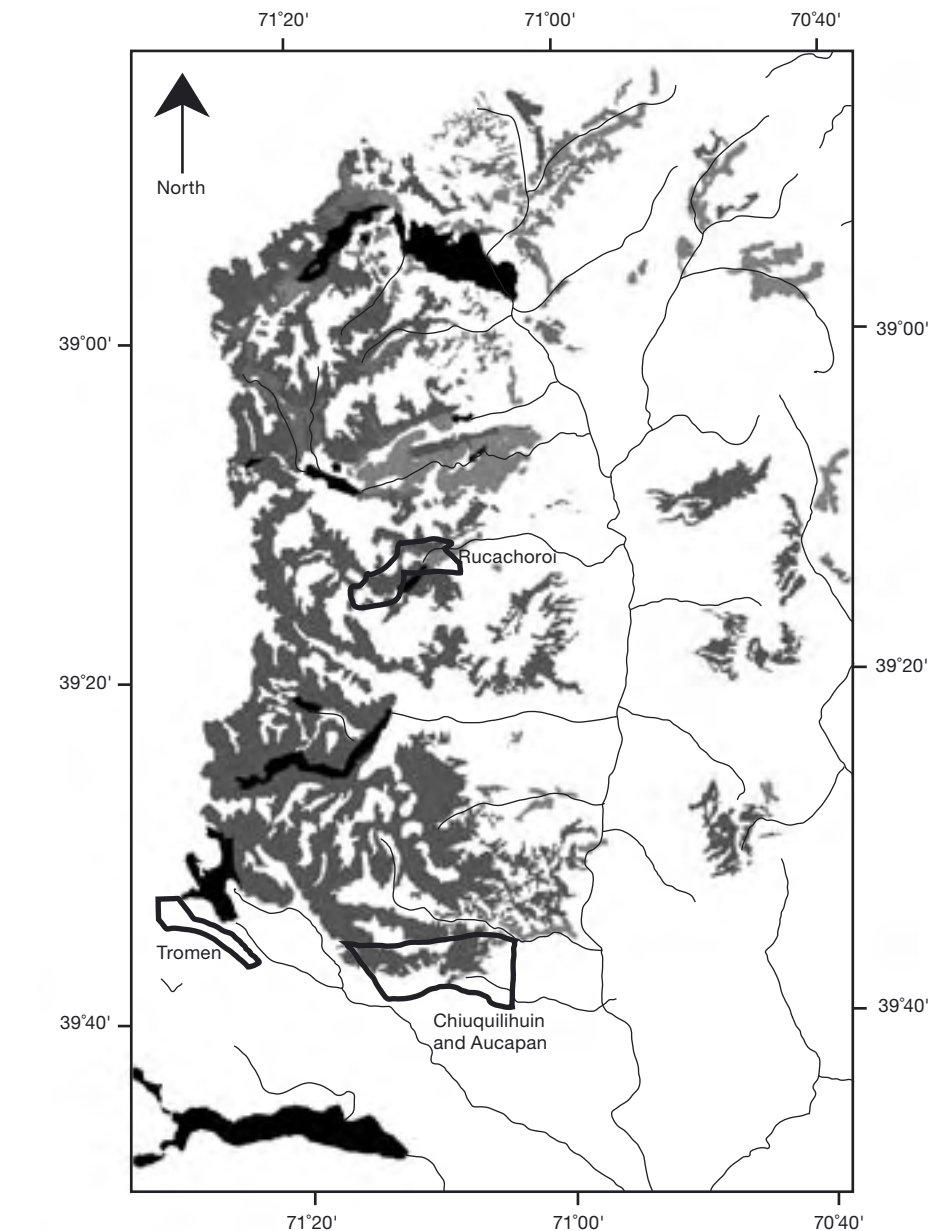


Figure 1. Map of the Argentinean range of *A. araucana* (in grey), and location of four study sites (Chiuquilihuin, Aucapan, Tromen and Rucachoroi). Source: Laboratorio de Teledetección – INTA Bariloche, Argentina.

Box 1. Araucaria and people in Argentina

Araucaria or pehuén has both religious and economic significance for the 60 000 Mapuche people (*mapu* = earth, *che* = people) living in the southern Andes mountains, and it has given its name to one of the Mapuche tribes: the Pehuéñches. Indigenous livelihoods are derived from livestock rearing, agriculture, production of wooden and woollen handcrafts, and different types of off-farm income such as government subsidies, salaries and pensions. From the pehuén forest, people obtain fuelwood, building materials both for their houses and shelters for livestock, resin for medicinal purposes, and food in the form of piñon. Piñon consumption varies in each household with seeds consumed as food, fed to livestock and sold as surplus. In some Mapuche communities, pehuén seeds represent from 10% to 15% of the diet during harvest time (February–May) and during the long winter (June–September) when they provide the main carbohydrate source. Tonnes of seeds are collected every year as food, and in the past some were exported. However, in 2002 araucaria was included in Appendix I of the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES) and the international commercialization of seeds from Argentina and Chile was prohibited. Inside Argentina's Lanin National Park indigenous people are allowed to use dead wood for fuel and carvings, and to collect up to 300 kg of seeds per family per year. For some families, especially in Chile, the sale of araucaria seeds in local and national markets provides an important annual income.

2.1 Threats

About 60% of the Argentine araucaria forests remain after deforestation (Veblen *et al.* 1999; Rechene 2000), and these continue to be degraded by fire, logging and overgrazing. Today, the most obvious sign of araucaria forest degradation is the lack of natural regeneration that, under normal conditions, follows a 'pulse' pattern of highly productive seed years followed by less productive ones. Livestock and wild animals also exert pressure on araucaria stands, consuming seeds in autumn and seedlings in spring, and trampling seedlings during grazing. Non-native mammals such as red deer (*Cervus elaphus*), wild boar (*Sus scrofa*), European hare (*Lepus europaeus*), European rabbit (*Oryctolagus cuniculus*) and Norway rat (*Rattus norvegicus*) have invaded forests where livestock are absent, with serious detrimental effects on native forest ecosystems (Funes 1996; Flueck *et al.* 1999). In forests experiencing intensive human use in the form of seed collecting and animal grazing, araucaria trees also exhibit low regeneration, and any regeneration that does occur is principally asexual with trees sprouting from roots (Figure 2, overleaf). In heavily used araucaria forests in Argentina and Chile, even trees that are over 80 years old can remain stunted (Veblen 1982; Rechene 2000).

3. Project objectives

The use and management of araucaria genetic resources, and the effect that humans and the environment have on the genetic diversity of the species, were examined within the framework of this project, jointly run by the International Plant Genetic Resources Institute (IPGRI) and the BMZ. Socioeconomic surveys were carried out in two Mapuche communities in Argentina to document current use patterns of araucaria FGRs. Araucaria genetic diversity was investigated within the natural range of the species in Argentina and also at the local level in the southern part of its range within the territories of two indigenous communities near the Lanin National Park (Figures 1 and 8), as well as in two forest stands, Rucachoroi and Tromen, the latter only partly located within the Lanin National Park. Seed productivity and predation studies were undertaken to serve as a basis for the later interpretation of observed regeneration patterns. Research results were



Figure 2. Asexual regeneration in *Araucaria araucana*, re-sprouting from a stump (photo: B. Vinceti).



Figure 3. A member of the Mapuche community of Aucapan, leading sheep to pastureland (photo: A. Pinna).

translated into practical recommendations for the management of natural resources within the areas studied, including the conservation of araucaria FGRs.

3.1 The study sites

Our project focused primarily on the Mapuche communities of Chiuquilihuin, where araucaria forest use was moderate, and Aucapan (Figure 3), where forest use was intense. These communities are located in Argentina's Neuquén Province in Andean Patagonia, between approximately 39°35'30"S, 71°13'30"W and 39°39'30"S, 71°0'0"W and include araucaria stands which are located in the most southern part of the natural range of *A. araucana* in Argentina.

These communities are bordered by Lanin National Park to the north and west and by private ranches to the south and east (see Figure 8, on p. 119). The topography is characterized by hills and mountains that vary from 750 to 2000 m above sea level (m asl), with average annual precipitation ranging from 1800 to 1200 mm from west to east.

A third area was studied (Tromen) that is partly contained within the boundaries of Lanin National Park and partly defined as forest reserve and under a different management regime. The part of Tromen located inside the national park was adopted as a control because it is less affected by human use. Finally, the study also included araucaria stands in a fourth site with higher precipitation, located in the Rucachoroi watershed north of the Chiuquilihuin and Aucapan communities, (39°14'0"S, 71°10'0"W), where signs of intense araucaria forest use were recorded. Average annual precipitation there ranges from 2200 to 1800 mm from west to east, with the altitude varying from 1350 to 2300 m asl. The watersheds studied all have a prevalent west–east orientation.

Forest associations of araucaria and *Nothofagus* species were characteristic of both community territories. Box 2 (overleaf) explains the main characteristics of the Andean forests and of the tree species associations found at the study sites (see also Section 4.1 in this chapter).

Large areas were cleared after European colonization, causing a drastic reduction in the natural range of araucaria forest. In Argentina, the establishment of a National Park Administration in 1937 represented the first step towards the conservation of native forests.

3.2 Demographic and socioeconomic aspects of the Mapuche communities

The Mapuche communities use the forest to gather seeds and fuelwood and to graze their domesticated animals at varying degrees of intensity. The forest is degraded by overgrazing, especially towards the eastern steppe which occupies the drier end of the rainfall gradient covered by araucaria stands.

The Chiuquilihuin and Aucapan communities have official legal rights to the forest and surrounding lands. Houses are distributed within the community territories, but most are concentrated in the drier steppe. The grazing areas are not clearly demarcated. The communities have their own statutes and representative bodies such as 'Directive Commissions' that are elected by secret ballot every two years. The commissions are headed by a 'cacique or lonko' who is often a spiritual leader in the community. Important issues are decided at community meetings.

The two communities surveyed, Aucapan and Chiuquilihuin, exhibited distinctly different levels of involvement in the project. Previous negative experiences of collaboration with research projects made the process of trust building slow and difficult, especially in the community of Aucapan where the data ultimately collected were much less complete.

Despite these problems, socioeconomic surveys were carried out using a participatory approach. The experience and knowledge of the indigenous people quickly revealed itself to be crucial to the reconstruction of past practices and to the identification of land-use options. Indigenous community surveys included: (i) demographic analyses; (ii) land-use map updates; and (iii) documentation and analyses of economic activities.

Box 2. The Andean woodland of Patagonia (Bosque Andino Patagónico)

The Andean forests of Patagonia extend from latitudes of 35° to 55° south on both sides of the Andean cordillera. They are defined as 'temperate forests' because of this location and because of the low winter temperature tolerance shown by the woody species of which they are primarily composed. To the east they border the Patagonian steppe, to the south the Atlantic Ocean, to the west the Pacific Ocean and to the north the Atacama desert. They thus form a relatively isolated forest biome and biogeographic island in southern South America (Armesto *et al.* 1995).

The Andean forests of Patagonia are dominated by broad-leaved species in association with a few conifers such as *Araucaria araucana*, *Fitzroya cupressoides* and *Austrocedrus chilensis*, and they are characterized by greater species diversity and higher species endemism than forests in the equivalent climatic zone in the northern hemisphere, indicating a distinctly different origin for the southern flora (namely Gondwana; Arroyo *et al.* 1995). Of the broad-leaved species, ten are found in the genus *Nothofagus*, and these species constitute the most important component of the Andean forests of Patagonia. They are found in a wide range of habitats, showing both great phenotypic plasticity and genetic differentiation over their natural range (Donoso 1987). *A. araucana* is found mixed with these and other species along a precipitation and altitudinal gradient. The three most common associations are:

(i) *A. araucana* and *Nothofagus pumilio* (called 'lenga' in the Mapuche language). This association generally occurs in the western, wetter parts of the araucaria range at higher elevations (1100–1800 m asl) and on south-facing slopes. This forest type has been commercially exploited.

(ii) *A. araucana* and *Nothofagus antarctica* in shrub form (called 'ñire' in the Mapuche language). This association is found in east–west running valleys near the upper forest limit. The araucaria canopy emerges from a prostrate and shrubby layer of ñire. Currently, this forest type is an important source of fuelwood for Mapuche communities and is affected by grazing and by natural and/or man-made fires.

(iii) *A. araucana* in pure stands is found near the upper forest limit on poorer soils but also at lower elevations within the steppe environment. The isolated araucaria woodlots in the steppe are highly disturbed. The vegetation of this steppe ecoregion is xerophytic and highly adapted for protection against drought, wind and herbivores. Dwarf and cushion shrubs are the most widely occurring vegetation types in the ecoregion. There are three main types of vegetative communities: semi-desert (accounting for 45%), shrub-steppe (30%) and grass-steppe (20%). Desert-like areas also exist with little to no vegetative cover as do wet meadow areas which have close to 100% cover.

The Andean forests of Patagonia have been heavily exploited in the past, especially so in Chile.

Demographic analyses

Demographic surveys of the two communities were undertaken (Table 1). The family group was defined as the people who live in the same house and share costs. For Chiuquilihuin, the survey started with a list of families provided by the Neuquén Province food aid programme. Information was cross-checked with members of the Directive Commission and other members of the community, and official sources were also consulted. Less-detailed information was informally made available for Aucapan by 'Artesanías Neuquinas', a governmental institution providing handicraft marketing expertise and other services to the indigenous communities of Neuquén Province. Additional data on Aucapan were also available from a partial census carried out in March 2000 by a local nongovernmental organization (NGO), Cruzada Patagónica.

Table 1. Comparison of demographic data for Chiuquilihuin and Aucapan, Argentina.

	Chiuquilihuin			Aucapan		
	Source 1 (1987/88)	Source 2 (1988)	Source 3 (Jan. 2002)	Source 1 (1987/88)	Source 2 (1988)	Source 3 (Jan. 2002)
Family groups	37	55	68	80	120/100	183
Inhabitants	212	330	306	554	720/600	700 (approx)
Men	52.9%			53.3%		
Women	47.1%			46.7%		

Three sources were used to derive census data. In Chiuquilihuin, Source 1 = Census Social Welfare Ministry 1987/1988, Source 2 = National Agricultural Census 1988, Source 3 = Field census January 2002. In Aucapan, Source 1 = Census Social Welfare Ministry 1987/1988, Source 2 = National Agricultural Census 1988 (two different census figures are reported in different publications of the National Agricultural Census), Source 3 = data from local NGO Cruzada Patagónica regarding only 15 families but believed to be representative of the composition of the whole community. These data integrate information on Aucapan informally provided by Artesanías Neuquinas.

Gender distribution in the two communities was analyzed. The data derived from official sources were not cross-checked with members of the community during the BMZ project, but a new census was undertaken in January 2002 in order to gauge growth trends. These data, though limited, reveal substantial growth during the last 15 years.

Land-use map updates

Land-use maps were updated for the two communities, with the objective of locating the houses on community-controlled land. Information collected from local people and from recent satellite maps allowed us to create updated maps supported by geographic information systems (GIS), which were then delivered to the Directive Commission of each community. In Chiuquilihuin, houses were concentrated mostly along the Chiuquilihuin River within a limited geographic area, while Aucapan houses were more uniformly scattered across the community territory (Figure 4, overleaf). The extension and degree of fragmentation of araucaria forest stands within the communities' territory were assessed.

Documentation and analyses of economic activities

The socioeconomic survey of Chiuquilihuin enabled us to identify different categories of farmers on the basis of their income structure (Figure 5, overleaf). Including children and elders, the survey revealed that approximately 40% of the members of this community (representing approximately 90% of the households) receive off-farm income. While men and women have equal opportunities to obtain subsidies, men are favoured when looking for employment, most commonly as shepherds on private farms in neighbouring areas.

Provincial forest authorities also provide subsidies to farmers for planting trees on their land, and in 2002 the Neuquén provincial government provided a subsidy of US\$0.27 per animal per year to promote grazing in rural areas.

Within the two Mapuche communities, livestock overgrazing is a major cause of soil erosion and a factor that impedes tree regeneration. Therefore, a survey of livestock was undertaken to determine the number of animals owned within the community and the distribution of animal ownership across community members. Different sources were used to assemble these data, shown in Table 2 (on p. 113). The field observations that we made in February 2002, when compared with previous censuses, led us to conclude that prior censuses were serious underestimates.



Figure 4. A typical Mapuche house in the community of Aucapan (photo: L. Gallo).

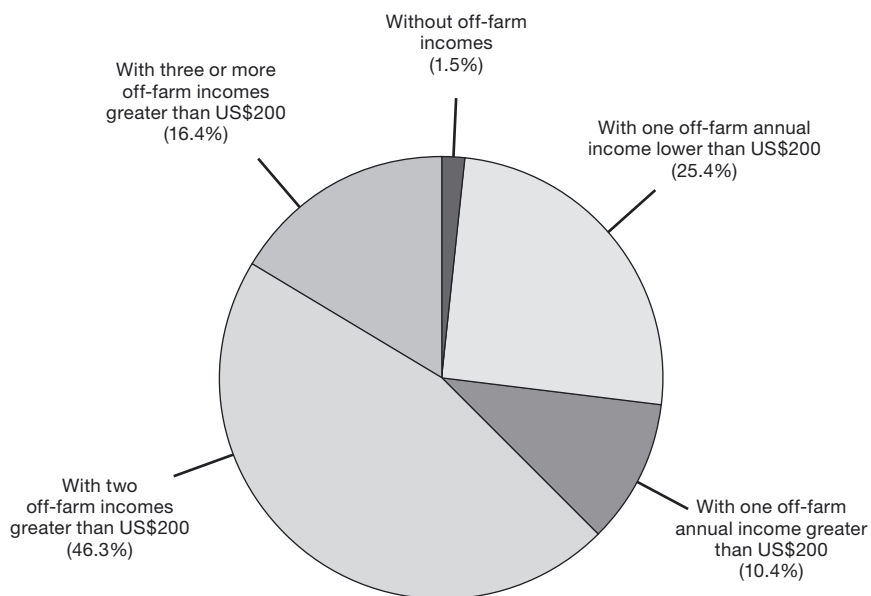


Figure 5. Composition and amount of off-farm income for different categories of families identified in Chiuquilihuin.

Table 2. Comparison of livestock in Chiuquilihuin and Aucapan, from three sources.

Chiuquilihuin											
Agricultural Department (Sept 2001)				Vaccination records (Nov 2001)			Chiuquilihuin community (Feb 2002)				
Sheep	Goats	Cattle	Horses	Sheep	Goats	Cattle	Sheep	Goats	Cattle	Horses	
654	721	312	62	592	409	100	774	873	450	148	
37.4%	41.2%	17.8%	3.5%	53.8%	37.1%	9.1%	34.5%	38.9%	20%	6.6%	
Aucapan											
Agricultural Department (Sep 2001)											
Sheep	Goats	Cattle	Horses								
1 467	853	175	150								
55.5%	32.2%	6.6%	5.7%								

Sources: the Agricultural Department of Neuquén Province (Sept 2001), the Neuquén Province veterinary service (Nov 2001), and the Directive Commission and various farmers from Chiuquilihuin (Feb 2002). For Aucapan, only data from the Agricultural Department (September 2001) were available.

This information allowed us to estimate the overall grazing pressure of animals both on pastureland and on the grazed forest area within the two community territories (see Section 4.4 in this chapter). According to the size of their herds, Chiuquilihuin farmers could be divided into four groups of livestock ownership as shown in Figure 6 (overleaf). Most community members owned small numbers of animals. Large-scale farmers were more likely to own cows, while small- and medium-scale farmers were more likely to own sheep and goats.

As discussed later in Chapter 12, we employed a participatory approach to carrying out the socioeconomic survey. Despite problems related to what proved to be the time-demanding and slow process of building trust between the researchers and the local communities, it was possible to collect a large amount of information on indigenous forest resource use, livestock management and other economic activities. These data turned out to be crucially important in our attempts to address some of the practical issues raised by the Mapuche communities involved in this BMZ subproject (see Section 6 of this chapter).

4. Description of the araucaria forest stands studied

4.1 Forest types, distribution and dynamics

The forest stands that we studied near Tromen and Rucachoroi are both located within Lanin National Park but differ from one another in their composition, density, structure and intensity of human use (Tables 3 and 4, on p. 115), with the site in Rucachoroi showing more signs of human activity. The location of the forest stands investigated is shown in Figure 1. Tree density and basal area calculations helped us to identify the structural conditions and the developmental stages of different stands within Tromen and Rucachoroi forests (Table 3).

In Tromen, a significant difference was found between western and eastern stands. The western stand was in a more advanced developmental stage, while the eastern stand

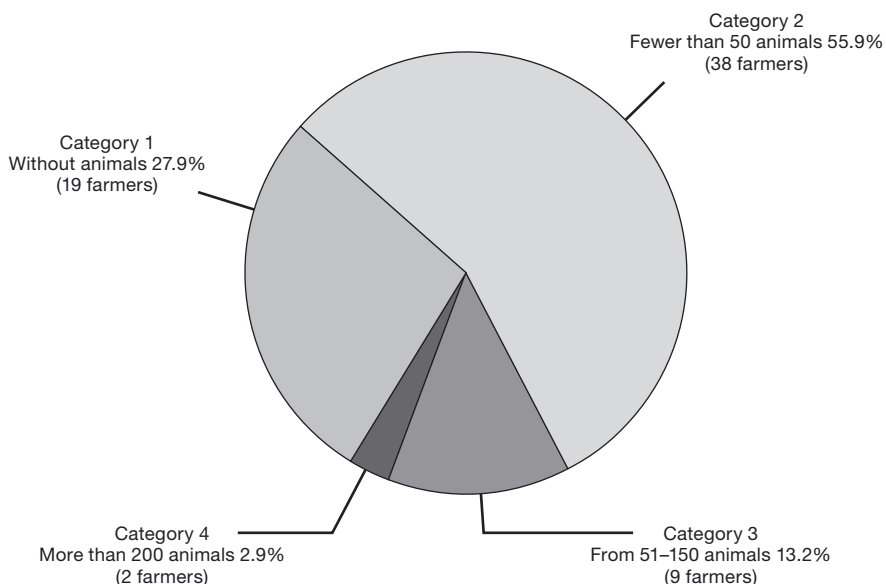


Figure 6. Livestock ownership in the Chiuquilihuin community in 2002. No farmers owned between 151 and 199 animals.

showed a bimodal pattern, indicating disturbance probably either by fire or human activity within the last 200 years. The level of regeneration in the two stands was substantially different. More than 24 000 seedlings/ha were found in the western stand, while fewer than 1700 seedlings/ha were found in the eastern stand.

In Rucachoroi, to account for the larger range of forest types, we divided forest stands into four types according to location, composition and structure (Table 4): dense araucaria (Aa) stands in the east; Aa in the west; open araucaria + *Nothofagus pumilio* mixed stands (Np); and, araucaria + *Nothofagus antarctica* (Na). The level of regeneration in the four types of Rucachoroi stands was low (<1700 seedlings/ha), most likely as a result of historical livestock overgrazing (Rechene 2000). Larger proportions (24–57%) of trees with umbrella-like shapes were found in the dense stands. In contrast, open stands had a greater proportion of trees with conical shapes (47–82%), typical of individual trees that have been growing without competition from neighbouring individuals.

4.2 Patterns of seed production

In 2000, araucaria seed production in Lanin National Park was studied across the range of environments. Six forest stands were selected based on forest structure and position in the precipitation gradient. Stands were chosen using a stratified sample design based on work performed by Burns (1991) and aerial photographs (scale 1:28 000). The stands were characterized with regard to overall tree density, density of seed trees, tree size (shape, height and dbh) and sexual information, using plot methods (point-centred quarter method) applied on line transects with different geographic orientations (Mateucci and Colma 1982; Sutherland 1996). Sample points were located every 40 m in dense forest and 60 m in open forest.

Seed production estimates during 2000, 2001 and 2002 were performed visually using binoculars, by counting cones on seed trees randomly sampled within defined size classes. Assuming similar production for sampled sites and seemingly identical sites not sampled,

Table 3. General characteristics of stands studied inside Lanin National Park.

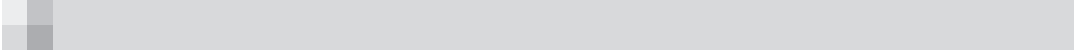
dbh class (cm)	Open Tromen		Dense Tromen		Open Rucachoroi		Dense Rucachoroi	
	Trees/ha	BA (m ² /ha)	Trees/ha	BA (m ² /ha)	Trees/ha	BA (m ² /ha)	Trees/ha	BA (m ² /ha)
10	16.2	0.2	82.1	1.3	21.0	0.4	82.0	1.3
20	21.5	1.0	23.8	1.2	16.6	0.8	24.0	1.1
30	30.0	2.8	23.8	2.4	10.0	0.8	22.0	2.0
40	24.2	3.9	19.5	3.2	9.1	1.4	8.0	1.4
50	5.4	1.4	23.8	5.7	5.8	1.4	16.0	3.8
60	5.4	1.7	13.0	4.4	8.3	2.6	16.0	5.1
70	0.0	0.0	10.8	4.4	4.2	1.7	16.0	7.2
80	8.0	4.4	13.0	7.7	4.2	2.3	2.0	1.1
90	0.0	0.0	4.3	2.1	0.8	0.6	4.0	2.8
100	0.0	0.0	0.0	0.0	0.8	0.7	2.0	1.8
110	0.0	0.0	4.3	3.5	1.7	1.7	6.0	6.2
120	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
130	0.0	0.0	0.0	0.0	0.8	1.2	2.0	3.0
140	2.7	4.4	0.0	0.0	0.0	0.0	0.0	0.0
Total	113.4	19.8	218.4	35.9	83.3	15.6	200	36.8

dbh = diameter at breast height, BA= basal area expressed as m²/ha.

Table 4. General characteristics of the stands studied inside Lanin National Park. Standard errors in parentheses.

Forest Type	Location	Association of species	Average no. of trees/ha (dbh >10cm)	Sex ratio (%)			Seed trees/ha
				M	F	ND	
<i>Rucachoroi</i>							
East open forest	39°14'0"S 71°10'0"W	<i>Aa</i> pure	83 (13) n = 25	42	26	32	22 (3)
West open forest	39°14'0"S 71°15'0"W	<i>Aa</i> and <i>Np</i>	50 (21) n = 5	43	30	27	15 (6)
East dense forest	39°14'06"S 71°10'41"W	<i>Aa</i> and <i>Na</i>	200 (36) n = 22	29	24	47	48 (9)
West dense forest	39°15'01"S 71°12'58"W	<i>Aa</i> pure	253 (69) n = 10	36	13	51	33 (9)
<i>Tromen</i>							
East open forest	39°36'54"S 71°20'28"W	<i>Aa</i> and <i>Na</i> or <i>Np</i>	113 (23) n = 10	24	31	43	35 (7)
West dense forest	39°34'59"S 71°27'41"W	<i>Aa</i> and <i>Np</i>	216 (41) n = 17	18	24	58	52 (10)

n = number of sites sampled, *Aa* = dense araucaria stand, *Np* = araucaria + *Nothofagus pumilio*, *Na* = araucaria + *Nothofagus antarctica*. M = male, F = female, ND = not determined.



gross araucaria seed production for the Tromen and Rucachoroi watersheds (western and eastern parts) was estimated. In Rucachoroi, we compared seed production estimates with official seed extraction data per hectare and concluded that when seed production is high, seed collection by local people also increases. Seeds are more easily harvested in productive years because people spend less time searching for suitable collecting sites. Thus, in order to estimate the historical periodicity of high seed production, we used data on seed collecting by humans between 1987 and 2000 in Rucachoroi as well as from several sites in Neuquén Province about 150 km to the northeast of Rucachoroi. Overall seed production for the six sampled stands was monitored over three years (Table 5).

In general, dense araucaria forests in Lanin National Park were more productive than the open forests, with the exception of the open and dense western stands in Tromen, where this pattern was reversed (Table 5). In the very dense Tromen stand (see Table 4), competition among seed trees growing close together with typical, smaller umbrella-like crowns, results in a lower productivity of seeds per hectare.

The production of araucaria cones per seed tree varied greatly from year to year. We estimate that the forests in the study area produce good quantities of seed every three years or so. Following high production years, seed trees require at least two years to recover, during which time seed production is lower. In the eastern, drier locations, seed production shows less annual fluctuation with precipitation, while in the wetter, western locations seed production is heavily depressed in years with high rainfall. This pattern may be due to the limited wind dispersal of pollen during years of heavy rain, which lowers seed output particularly in forests in the more humid areas located in western sites (Donoso 1998; Sanguinetti *et al.* 2000). This information strongly suggests that the weather influences araucaria seed production at the study sites and over the natural araucaria range in general.

4.3 Threats to araucaria forests from seed predation

Mapuche collect seeds for their own consumption, to feed livestock, and occasionally to sell to traders, but most of the pressure on araucaria regeneration comes from seed predation by livestock and wild animals in the fall, from browsing of seedlings by these same animals in the spring, and from their trampling of seedlings and saplings during both seasons.

Using microhistological analyses of samples of faeces, prepared and analyzed following the procedure of Holechek and Gross (1982), we investigated the proportion of araucaria seed in the diet of herbivores. Faecal samples from horses, cows, sheep, goats,

Table 5. Average range of seed production in forest stands studied inside Lanin National Park over a three-year period (2000–2002).

Stands (trees/ha, with dbh > 10 cm)	Seed productivity (kg/ha)	Forest area (approx. ha)	Total annual seed production (tonnes)
Rucachoroi			
Open eastern forest (<90)	100–200	500	50–100
Dense eastern forest (>200)	200–300	250	50–80
Open western forest (<90)	50–140	200	10–30
Dense western forest (>200)	100–160	300	30–45
Tromen			
Open eastern forest (<90)	200–300	50	10–20
Dense western forest (>200)	50–300	10	0.5–4

deer and wild boar were collected from winter grazing lands (invernadas) and summer grazing lands (veranadas) in continuous and fragmented forests located within Aucapan and Chiuquilihuin community territories, and at Rucachoroi and Tromen in late spring (December) 2000 and in autumn (April) 2001. Fenced enclosures and traps placed inside Rucachoroi and Tromen forests allowed us to monitor the effects of grazing and seed consumption by both livestock and wild species, including birds and rodents.

Despite the overall high digestibility of araucaria seeds, seed tissue residues were detected in most of the faeces examined from domesticated and wild exotic herbivorous mammals. Each of the herbivores' diets varied depending on the grazing habits of the different species and on the area studied (Figure 7). Araucaria seeds constituted a substantial part of the diet of two wild exotic herbivores (red deer and wild boar) and of cows.

Mapuche collect, on average, around 14% of total seed production (calculating a mean over high and low years of seed production), which affects regeneration of araucaria (Sanguinetti *et al.* 2002), but the consumption of seeds by animals is generally much higher. Wild exotic animals alone usually consume around 22% of overall seed production.

Our results show the need for greater control of access to forest stands by livestock and wild animals where regeneration of araucaria is threatened by predation and trampling of seedlings and saplings.

4.4 Threats from livestock overgrazing and a proposal for improved management

We undertook a study to determine sustainable concentrations of grazing animals for the different types of pasturelands found within the Mapuche communities of Chiuquilihuin and Aucapan, to see if those grazing areas could provide enough food for animals without the contribution of araucaria seeds. The objective of this analysis was to assess the possibility of formulating pasture management strategies that could optimize forage production and meet the food demands of livestock and wild animals while reducing the pressure on araucaria regeneration.

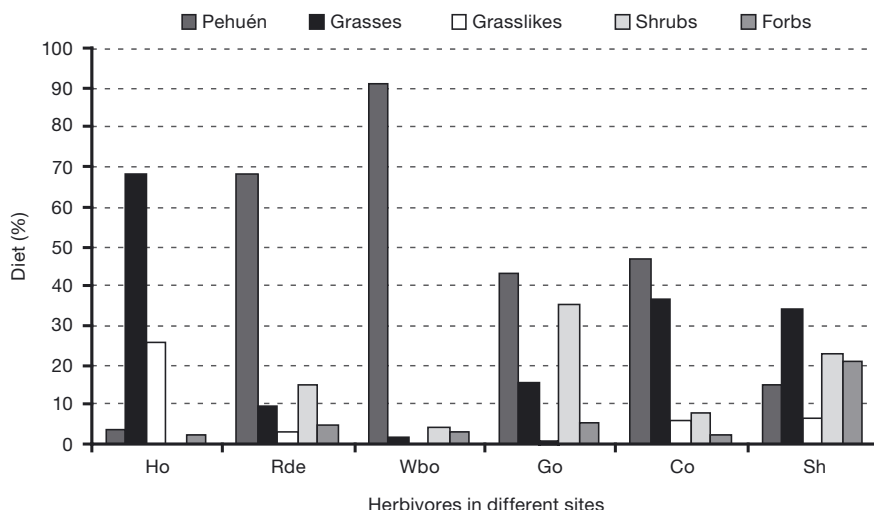


Figure 7. Composition of herbivores' diets in autumn, when they contain araucaria seeds. Ho = horse, Rde = red deer, Wbo = wild boar, Go = goats, Co = cow, Sh = sheep. Forbs: non-grassy herbaceous species, e.g., legumes and composites.

We distinguished pastureland types by the quality and the amount of forage each produced, using 1:60 000 maps derived from Landsat 7 TM images from 1999, images interpreted by the INTA GIS Laboratory in Bariloche, Argentina. Of the total land area studied in the two communities (14 114 ha), 63% was suitable for grazing. The different types of pastures identified, based on different plant assemblages, are described in Table 6.

For each type of pastureland identified in Table 6, we determined the actual and recommended animal stocks in Sheep Livestock Units (SLUs) per ha, using a standard Merino sheep of 40 kg, fed with 365 kg of dry matter/year. This was done on the basis of forage produced by each pasture type, its total extension, its accessibility to livestock, and a coefficient we called 'use factor' (Stoddart *et al.*, 1975) that relates to the extent of degradation of the vegetation and soil. Household surveys showed that the number

Table 6. The principal pastureland types suitable for grazing, identified by INTA researchers on the basis of plant assemblages, indicating environmental conditions and degree of degradation.

Type	Area	Characteristics
Wet meadow	69 ha	Located between 1200 and 1500 m asl. The principal species are <i>Juncus balticus</i> , <i>Poa pratensis</i> , <i>Hordeum</i> spp. and <i>Trifolium repens</i> . Growth occurs during the summer months (December–March), with annual production between 3000 and 5000 kg of dry matter/ha. Wet meadows present moderate to high forage production, with a maximum during the summer.
Meadow	163 ha	Found near wet meadows. Currently intensely grazed, meadows annually produce between 1000 and 3000 kg of dry matter/ha. Meadows present moderate forage production and limited signs of degradation.
Low shrub steppe	2 758 ha	Found on sandy soils, between 1300 and 1500 m asl, this pasture type is characterized by two main vegetation layers: low shrub (0.5–1 m high) and grass (5–20 cm high). The principal species are <i>Mulinum spinosum</i> , <i>Senecio bracteolatus</i> , <i>Festuca pallescens</i> and <i>Stipa speciosa</i> var. <i>major</i> . Annual production is 100 to 200 kg of dry matter/ha. The key forage is <i>Festuca pallescens</i> (1–10 % of cover). The growing season is from December to March. Soils are heavily eroded and the vegetation is degraded.
Medium shrub steppe	2 421 ha	Found on sandy soils between 1200 and 1400 m asl, with the following dominant species: <i>Mulinum spinosum</i> , <i>Senecio bracteolatus</i> and <i>Festuca pallescens</i> . Two vegetation layers are characteristic: shrubs (10–30% of cover) and grass (5–10% of cover). Medium shrub steppe shows high grazing pressure, signalled by dominance of the annual species <i>Bromus tectorum</i> . The principal forage species are <i>Festuca pallescens</i> , <i>Poa lanuginosa</i> and <i>Bromus setifolius</i> . Most plant growth occurs in spring (September - December). Annual production is between 50 and 100 kg of dry matter/ha. Soils are seriously eroded and the vegetation highly degraded.
Dwarf shrub steppe	3 454 ha	Between 1000 and 1400 m asl, in areas likely occupied historically by prostrate <i>Nothofagus antarctica</i> . This formation is found in areas that are susceptible to water and wind erosion. The principal vegetation layer is a dwarf shrubby cover of <i>Acaena splendens</i> , <i>Adesmia boronioides</i> , <i>Mulinum spinosum</i> and <i>Berberis heterophylla</i> . Other grass species are <i>Festuca pallescens</i> and <i>Stipa speciosa</i> var. <i>major</i> (1–5% of cover). This pasture type's peak annual production is at the end of spring (November–December) and is low (20–50 kg of dry matter/ha). Soil is highly eroded and the vegetation is highly degraded.

of animals owned by the communities of Chiuquilihuin and Aucapan and their grazing density on pastureland were on average five times higher than our recommended density. Furthermore, only a few families own the most productive pasturelands and this impedes an optimal exploitation of these pastures during the summer.

Based on our analysis of forage available from different pastureland types, on the current size of the area grazed, and on the current rotation system used, we made recommendations for improved regulation of access to pasturelands, and proposed them to policy makers and local indigenous communities.

Suggested improvements in livestock management practices were based on changes to the number of animals grazed in summer pasturelands combined with the identification of new areas for forage harvesting and an increase in the area harvested for winter forage. However, there are serious constraints to applying any of these solutions, the most important being that a reduction in the number of livestock owned by the community would have a negative impact on a traditional, fundamental source of income. However, just a few families own a large number of animals, therefore a reduction of livestock (properly compensated) could be introduced at the expense of only a few individuals.

5. Forest genetic study

Considering the pressure posed by fragmentation, soil degradation, and lack of regeneration on araucaria genetic resources in Argentina, one of the objectives of our project was to survey both the ecological and genetic aspects of araucaria on a landscape scale in order to recommend best practices for the conservation and use of araucaria genetic resources. We were especially concerned by the state of the most fragmented araucaria stands located in the eastern, drier parts of the range in Argentina. Our study included both unused stands and those traditionally used by Mapuche indigenous communities within Lanín National Park and in surrounding areas (see Figure 8).

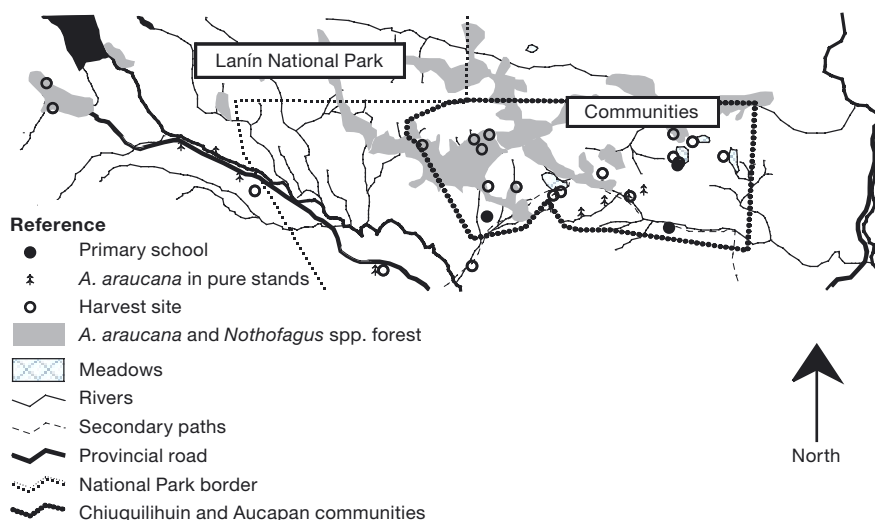


Figure 8. Boundaries of the two indigenous communities studied (Chiuquilihuin and Aucapan) and of Lanín National Park. The areas where Mapuche people collect araucaria seeds are indicated.

In order to monitor genetic diversity, describe genetic variation and study genetic processes, well-defined gene markers are needed. Isozyme markers were identified for this species, and progeny and provenance trials with araucaria were initiated. We also undertook genetic diversity and differentiation surveys of the species in its natural Argentinean range (Figure 9).

Isozyme markers were preferred for population genetic studies because, among their advantages, their co-dominant expression is especially important. Since endosperm tissue carries the information of two gametes, segregation patterns can be directly observed through the analysis of mega-gametophytes. Additionally, isozymes function as adaptive markers (Bergmann and Hattmer 1998) and this makes them useful for selection process analysis.

Our forest genetic study of *A. araucana* in Argentina occurred on two spatial scales. Research at the local level analyzed the southern natural distribution of the species in Argentina. The objective was to describe the genetic connectivity among fragmented and continuous araucaria neighbouring populations and the effect of human use on the genetic diversity of the species. A second study was carried out at a regional level, throughout the natural range of araucaria in Argentina. The objective was to describe the general pattern of variation of the species in that country.



Figure 9. Seedlings of *A. araucana* for progeny and provenance tests at the tree nursery of INTA, Bariloche, Argentina (photo: A. Jarvis).

5.1 Materials and methods

Determination of isozyme markers

Thirteen isozymes were assayed using laboratory protocols from Cheliak and Pitel (1985), slightly modified for *araucaria*. Sampling of trees occurred in four *A. araucana* forest fragments located in the eastern, drier steppe, where trees were widely separated and seeds on the ground could be assigned to a specific individual with reliability.

Putative heterozygous individuals were detected by screening nine randomly selected seeds per tree. Seeds from heterozygous trees were used to test the observed segregation against the 1:1 Mendelian theory of inheritance (Chi-square test, $\alpha = 0.05$), expressed by the law of segregation (each hereditary characteristic is controlled by two alleles, which segregate and pass into separate germ cells) and the law of independent assortment (the pairs of alleles segregate independently of each other when germ cells are formed). This allowed us to determine the gene markers.

Population analysis

Bulk stand collecting and individual tree collecting were undertaken to gather seeds from 22 populations covering the natural range of the species in Argentina during two consecutive years (March–April of 2000 and 2001). Samples were taken from inside the Chiquilihu community (moderate use of *araucaria*), in the Aucapan and Rucachoroi community lands (intensive use of *araucaria*), and in the Tromen forest (control area located inside Lanin National Park).

The selected populations corresponded with the three most common *araucaria* forest types: *Np*, *Na* and *Aa* (from Table 4). These associations integrate with each other in the order presented along the west–east decreasing precipitation gradient. Continuing eastwards, the forests progressively disappear and are replaced by nonforested steppe.

A local variation analysis was performed on several progenies originating from five fragmented but pure *araucaria* eastern populations (Figure 10) and nine additional populations from western mixed stands of *araucaria* + *Nothofagus* spp. A regional variation analysis of *araucaria* included 11 populations covering the natural range of the species in Argentina, including three that were analyzed in the local level analysis.

Determination of genetic parameters

Genetic diversity (variation within populations) and differentiation (variation between populations) were estimated in the Argentinean *araucaria* populations. For this purpose, 100 seed embryos from various populations spread within the *araucaria* range were analyzed electrophoretically. This sample size guaranteed the inclusion of alleles with a



Figure 10. Examples of dense and fragmented populations of *A. araucana* from the study sites (photo: B. Vinceti).

frequency value of 3.2%, and with a probability level of 0.95 (Gregorius 1980). For the analysis of genetic diversity, the parameters listed below were measured:

- Average number of alleles per locus (A)
- Gene pool diversity (ν) equivalent to the effective number of alleles (n_e)
- Observed heterozygosity (H_o)
- Expected heterozygosity (H_e)
- Fixation index (f), a measure of heterozygote deficiency or excess (Wright 1978).

For genetic differentiation, the parameters measured were:

- Homogeneity test-G (significant differences among populations structures)
- Absolute genetic distance (d_o ; Gregorius 1974)
- Gene pool differentiation of a population or deme (D_j ; Gregorius and Roberds 1986)
- Gene pool differentiation (δ ; Gregorius and Roberds 1986).

Significance of the genetic distance was inferred in terms of probabilities (through the matrix of probabilities $\alpha = 0.05$). F_{ST} is the most widespread measure of genetic differentiation used and describes the proportion of variance within a species that is due to population subdivision (Wright 1978). A cluster analysis based on the genetic distance proposed by Gregorius (1974) was adopted.

F_{ST} was used to obtain an indirect estimate of the historical number of migrants per generation (N_m ; Wright 1951).

5.2 Results of the forest genetic study

For the genetic analysis of the observed phenotypic segregation (shown as bands in the zymograms) 16 putative heterozygote trees were used. The genetic analysis proved that the observed variation in four polymorphic loci (*Idh*, *Pgm-A*, *Pgm-B* and *Got-C*) was genetically controlled. The results of the Chi-squared tests for single-locus segregation are presented in Table 7 (opposite). The probability values indicate that there are no significant differences between the observed and expected segregates. Therefore, we could statistically determine nine gene markers for this species, distributed in four gene marker loci. Other observed genetic segregating variants without a statistical verification were not included in the population genetic studies.

Table 8 (on p. 124) summarizes most of the genetic parameters estimated for local and regional Argentinean araucaria populations. For all populations, the average number of alleles per locus (A/l) was 2.5 (in each population all the possible allelic variants were detected). The mean gene pool diversity (ν) for the local level analysis was 1.820, a value significantly higher than the 1.694 mean gene pool diversity for the regional level data. The AE sample, from a marginal population of isolated trees in the heavily exploited Aucapan steppe, had the highest mean gene pool diversity of 1.998. The lowest value corresponded to RO, a Rucachoroi west population just north of the Aucapan and Chiuquihuín communities in a neighbouring valley that has a history of land use and forest fires.

The observed heterozygosity (H_o) is based on observed genotypes and, therefore, it better expresses a real condition of genotypic variation versus potential variation (expected heterozygosity H_e) and was similar in local and regional level analyses. H_e is the level of heterozygosity expected if the population were in Hardy-Weinberg (H-W) equilibrium (Nei 1973), which defines the genotypic frequencies expected when genetic drift is not occurring, mating is random, natural selection is taking place and neither migration nor mutations occur. The observed differences from these partly unrealistic conditions explain how Mendelian segregation influences allelic and genotypic frequencies in a population.

Significant deviations from H-W expectations were observed for the four loci analyzed (as shown in Table 7) in several populations. In most cases this was due to an excess of homozygotes that are associated with inbreeding. This pattern was most prevalent in the western, continuous populations where higher rainfall probably restricted anemophilous (by wind) gene flow and contributed to creating a marked spatial genetic structure in these populations. In fragmented populations, we attributed inbreeding to the fact that seed dispersal occurs by gravity.

Table 7. Genetic analysis of phenotypic isozyme segregation, observed vs. 1:1 expected, in four putative *gene loci* of *A. araucana*. Chi-square value for two classes, with $\alpha = 0.05$, is 2.84. Tree Code: AE = Aucapan estepa; ANL = Aucapan lenga; TE = Tromen estepa; NMA = Nahuel mapi abajo.

Tree Code	Locus	Putative genotype	Observed segregation ratio	Chi-square
AE-3	Idh	100/114	43:44	0.01
AE-4			53:39	2.13
AE-5			42:55	1.74
AE-9			43:43	0.00
ANL-13			23:23	0.00
TE-9	Pgm-A	103/96	15:13	0.14
AE-3			32:45	2.19
AE-6			21:19	0.09
AE-8			49:42	0.53
AE-4			43:51	0.68
NMA-1	Pgm-B	103/90	38:33	0.35
AE-3			31:44	2.25
AE-4			50:36	2.27
AE-8	Got-C	100/300	38:48	1.16
AE-6			16:19	0.25
AE-3			42:31	1.65

Genetic differentiation among populations was found to be moderate with F_{st} equal to 0.10 (Hartl and Clark 1997). A value of Gregorius' δ of 0.143 among populations is three times higher than in other Patagonian native forest species (e.g., *Austrocedrus chilensis*: $\delta = 0.043$, Pastorino *et al.* in press; *Nothofagus nervosa*: $\delta = 0.047$, Marchelli and Gallo 2001). Local level mean values were slightly lower than for those at the regional level ($F_{st} = 0.095$; $\delta = 0.143$).

The gene pool differentiation of each population is graphically presented in Figure 11 (on p. 125) in a so-called 'differentiation snail' (Gregorius and Roberds 1986). The length of the radius of each 'pie portion' denotes the population differentiation with respect to the rest of the populations (D_j), their angles represent the population sizes weighted (c_j ; considered to be equal in this analysis), and the radius of the circle is the gene pool differentiation ($\bar{\delta} = 0.143$). The RO population was the most differentiated ($D_j = 0.23$), with the lowest values of gene diversity (0.1453) and H_o (0.174), and minor polymorphism in the four analyzed loci. On the other hand, TE is the population that best represents the general gene pool constitution of the species in Argentina.

Since significant genetic differences among population structures were found, gene pool distances were calculated (Gregorius 1974) and are shown in a cluster analysis in Figure 12 (on p. 125). The significance of the gene pool distances among populations was then demonstrated using a matrix of probability. Most of the gene pool distances were significantly different from a pooled population without subdivisions.

The cluster analysis formed three well-defined groups of araucaria. The first consisted of mixed araucaria populations with *Nothofagus pumilio*, such as CP and TO located at Tromen in stands with low use intensity. Also in this genetic cluster were CHO, located within the Chiuquilihuin community territory and also with low use intensity, and RO and RE, located in the Rucachoroi basin to the north of Chiuquilihuin and Aucapan and with

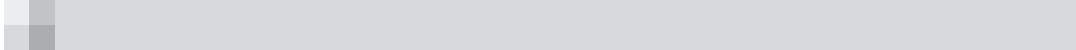


Table 8. Genetic diversity parameters of *A. araucana* populations.

Regional population samples	v	H_o	H_e	Dj (%)	f
PP – Primeros pinos (F)	1.564	0.183	0.360	12.8	0.457
PH – Pino hachado (C)	1.641	0.243	0.391	8.0	0.316
LL – Lonco loan (F)	1.703	0.185	0.413	22.7	0.430
RAL – Rio Aluminé (F)	1.689	0.228	0.408	7.7	0.445
CA – Caviahue (F)	1.888	0.244	0.470	18.8	0.424
LP – Lago Pulmari (C)	1.511	0.101	0.338	12.9	0.684
AL1 – Aucapan Lengua (C)	1.719	0.171	0.418	12.6	0.612
RE1 – Ruca este (F)	1.736	0.281	0.424	19.2	0.391
MQ – Moquehue (C)	1.741	0.192	0.426	9.6	0.432
TE1 – Tromen estepa (F)	1.714	0.312	0.417	14.0	0.264
PA – Paimun (C)	1.723	0.168	0.420	22.2	0.568
Local population samples					
AE – Aucapan estepa (F)	1.998	0.200	0.499	14.2	0.579
NMA – Nahuel mapi abajo (F)	1.803	0.217	0.446	18.3	0.512
ALN – Aucapan lenga (C)	1.708	0.204	0.415	17.0	0.528
TR-Ñ – Treguaricum ñire (C)	1.808	0.257	0.447	17.0	0.419
TR-L – Treguaricum lenga (C)	1.940	0.299	0.485	10.7	0.325
CHE – Chiuquihuín estepa (F)	1.880	0.208	0.468	13.9	0.485
RE – Ruca este (F)	1.753	0.265	0.430	11.4	0.384
MAU – Mauda-lenga(C)	1.903	0.282	0.475	11.9	0.372
CHO – Choroi-lenga (C)	1.866	0.244	0.464	12.5	0.471
APR – Aprovechado-lenga (C)	1.858	0.174	0.462	11.1	0.583
RO – Ruca oeste (C)	1.453	0.174	0.312	23.5	0.495
CP – Caña plantada (C)	1.865	0.187	0.464	13.0	0.559
TE – Tromen estepa (F)	1.919	0.205	0.479	9.0	0.465
TO – Tromen oeste (C)	1.732	0.293	0.423	16.2	0.293
Mean, local level analysis	1.820	0.229	0.448	14.3	0.462
Mean, regional analysis	1.694	0.210	0.408	14.6	0.457

v (n_e) = Genetic diversity (effective number of alleles). H_o = observed heterozygosity. H_e = expected heterozygosity. Dj = Gene pool differentiation in a population. f = fixation index. RE, TE, ALN = sampled in 2000 for the local level analysis. RE1, TE1, AL1 = the same populations sampled in 2001 for the regional level analysis. F = fragmented population. C = continuous population.

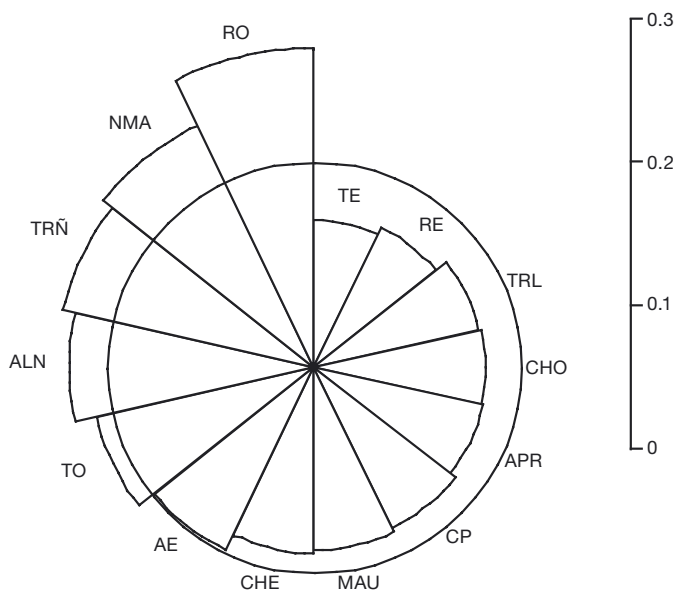


Figure 11. Allelic differentiation snail for the southern *A. araucana* populations (from the local level genetic study). For cited populations, see Table 8.

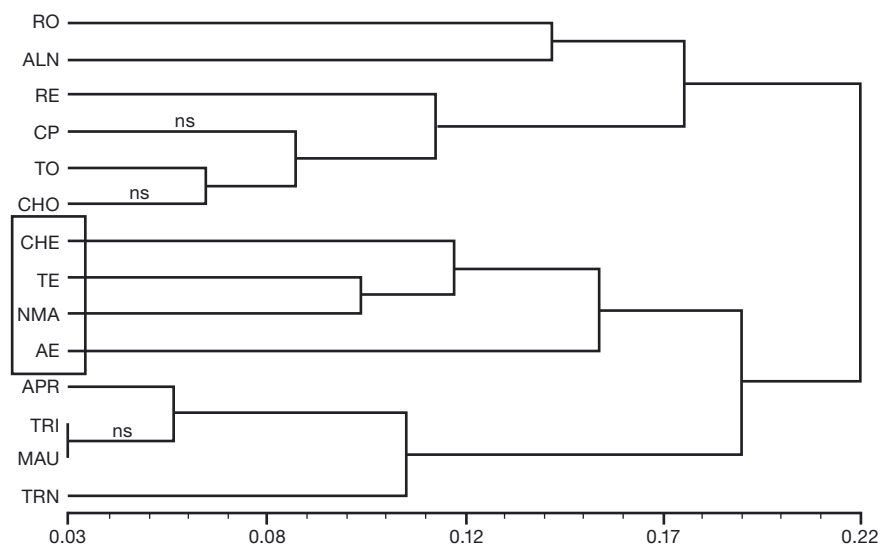


Figure 12. UPGMA (Unweighted Pair Group Method with Arithmetic mean) dendrogram for selected *A. araucana* populations. 'ns' means that the gene pool distance is not significantly different among populations. See Table 8 for populations cited on the y axis.

high use intensity. ALN, within Aucapan community territory and subjected to high use, also fell within this first cluster.

The second cluster was composed of several fragmented populations, CHE, TE, NMA and AE, that were generally located in areas marginal to the study site, at the southern and eastern limits of the araucaria range and in more arid climates. These populations were greatly affected by past and present human use and showed limited natural regeneration.

The last cluster consisted of four continuous populations located within the Mapuche communities of Aucapan and Chiuquilihuin: APR, TRL, MAU and TRN. These populations were characterized by moderate human use in the past (logging, fuelwood extraction, seed collecting and grazing of livestock) and currently exhibit different levels of natural regeneration.

Indirect estimates of the effective number of immigrants per generation were calculated (Wright 1951; Nei 1973) to characterize gene flow using all populations, resulting in a rather low number of $N_m = 2.22$. This value was lower than expected for gymnosperms when compared to the results from investigations carried out elsewhere on 89 species ($N_m = 3.17$, Hamrick *et al.* 1992) and lower than the values found for gene flow in some *Nothofagus* species (e.g., Premoli 1997; Table 9).

The low value of gene flow ($N_m = 2.22$) found may be due to the limited dispersal of araucaria's pollen and seeds. In addition, the prevailing wind, blowing from west to east during the pollination period, could be one of the factors responsible for the limited gene flow observed, with the western populations less likely to receive pollen from outside.

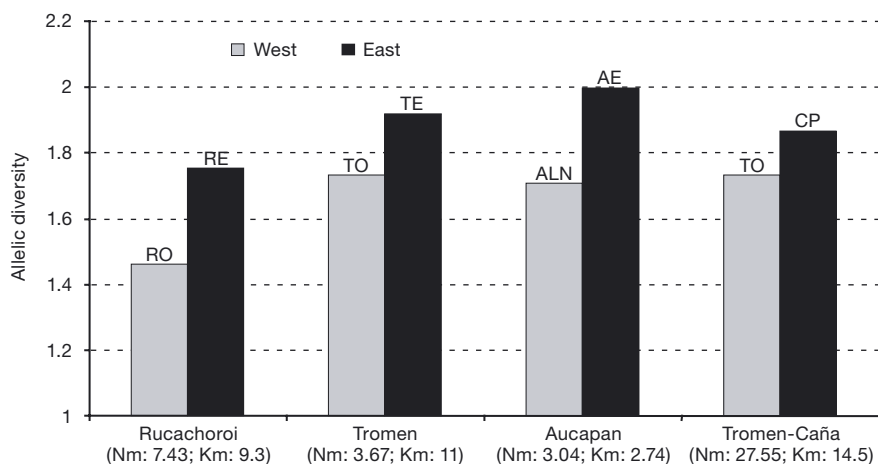
Gene flow was found to be slightly higher ($N_m = 2.39$) in the southern part of araucaria's natural range, where it was studied in a greater number of populations. However, the most pronounced differences in gene flow values within the range of the species were found when comparing samples from populations along west-east transects at similar latitudes within the same watershed. Similar patterns were found in neighbouring watersheds, supporting the theory of a strong unidirectional wind effect that induces pollen dispersion from west to east. In all cases, the estimates of gene flow were up to 12 times higher in the eastern stands. The genetic diversity found in most eastern populations was also higher (Figure 13).

The gene flow analysis replicated at a regional level confirmed the pattern found at the local level, showing eastern populations to have gene flow twice as high as in the western populations (Gallo 2003). We expected to see more genetic variation in years when more seeds were produced, implying more genetic contributions. Temporal mating system variation has been reported for *Nothofagus nervosa* species in relation to yearly variations in seed productivity (Marchelli and Gallo 1999), but this trend was not noted for araucaria during our study.

Table 9. Estimated number of immigrants per generation (N_m) for woody plants.

Category	Number of species	N_m
Gymnosperms*	89	3.17
Angiosperms**	102	2.20
Species		
<i>Nothofagus betuloides</i> **		1.80
<i>Nothofagus dombeyi</i> **		3.10
<i>Nothofagus nitida</i> **		5.00

* Hamrick *et al.* 1992, ** Premoli 1997



West-east population pairs and values of indirect gene flow estimations and distances

Figure 13. Relationship between allelic diversity (v) and indirect gene flow estimates (N_m) in four west-east population pairs from the southern natural range of *A. araucana*. Four pairs of populations were used for this analysis; the labels on the bars are the names of populations as shown in Table 8.

6. From genetic research to practical outcomes: the Pehuenche project

The information generated by the BMZ-funded project was used to implement a three-year rural development project called 'Pehuenche', which started in May 2003 with the financial support of the Italian Government's Ministry for Foreign Affairs, and coordinated by Angela Pinna for the Italian NGO Ricerca e Cooperazione, in collaboration with the Rural Development Department of INTA, Bariloche. This initiative was conceived following requests from Mapuche communities involved in the BMZ-funded project (Figure 14, overleaf).

The BMZ-funded subproject described here allowed us to gather baseline data that supported the work undertaken within the Pehuenche project, aimed at achieving the following objectives: (i) increasing household incomes through the rational use of natural resources and diversification of economic activities, (ii) solving technical water management problems, and (iii) strengthening the capacity of the indigenous communities to manage their own resources sustainably. The Pehuenche project sees Mapuche communities as active participants in all natural resource management decisions.

Besides being crucial to the promotion of local sustainable development initiatives, the Pehuenche project is also a promising model from which lessons of wide applicability can be drawn, with solutions potentially exported to other parts of the region where indigenous people hold communal land tenure rights within reserves. Indeed, the two Mapuche settlements involved in the Pehuenche project exemplify patterns of tension between traditional natural resource management regimes and the national and provincial governments seeking land for growing populations while dealing with natural resource use limits imposed by the proximity of a national park. Such local socioeconomic tensions are widely encountered in other rural regions of Argentina and in the South American countries generally.



Figure 14. Capacity building with Mapuche community members on livestock management inside Lanin National Park, as part of the activities of the Pehuenche project (photo: A. Pinna).

7. Conclusions

Our research allowed us to understand the patterns of forest resource use by several Mapuche indigenous communities located within the boundaries of Lanin National Park. Our study demonstrated the importance of araucaria forests to Mapuche people: seeds are used as food or to feed livestock and are occasionally sold on the market, and some araucaria forest stands are used for livestock grazing. The two communities studied, Aucapan and Chiuquilihuin, have different sizes, and forest stands included within their borders are subject to different use intensities. Livestock is a fundamental resource for Mapuche who are encouraged to raise more cows through subsidies allocated by the Neuquén provincial government to promote grazing in rural areas.

Nevertheless, our survey indicated that araucaria forests were subjected to heavy soil erosion caused by livestock overgrazing, and they show a lack of regeneration that is very likely caused by seed predation, livestock trampling of seedlings and saplings, and the resulting erosion. Cows and wild boars were found to be the largest consumers of seed among the species investigated. Without some sort of change, the current situation will become worse as the Mapuche population continues to grow.

Given this rather bleak future scenario, we believe that urgent measures are needed to improve or maintain forage production through adjustments in livestock numbers and grazing land management. The pasturelands used by the Mapuche communities were categorized into types according to their forage productivity and ecological characteristics, and this enabled us to estimate optimal animal stocks and densities for the different types of grazing sites, and these should be used as a basis for improving current livestock management practices.

Most of the solutions we envisage to improve livestock management imply a reduction of current animal stocks. The distribution of livestock ownership in Chiuquilihuin was

found to be highly unequal, with just two farmers owning most of the animals. Therefore, a significant reduction in the number of animals kept would affect only a very small number of people. Nevertheless, if this measure is to be implemented, targeted incentives should be provided to these community members to compensate them for their loss of income from livestock raising, and subsidies to promote grazing in rural areas should be stopped and replaced by different types of inputs (such as forage supplied in periods of scarcity, e.g., during winter).

The forest survey revealed that eastern araucaria stands are much more fragmented than western stands, and it showed slightly higher genetic fixation coefficients than the continuous ones. This could either be due to greater inbreeding or a reduced effective population size that could lead to genetic drift and increased homozygosity. Yet the eastern, more fragmented araucaria populations did not show worrying processes of genetic erosion when compared to western populations (Gallo 2003). However, in most of the fragmented populations of *A. araucana* studied, regeneration was absent or reproduction was mainly vegetative. Therefore, future adult generations in these fragmented populations will most likely show very different genetic structures from the ones observed today. The future of the fragmented stands located in the most arid parts of the range will most likely be characterized by serious reductions in genetic diversity.

No clear trends of diversity or differentiation in *A. araucana* populations were found in relation to environmental conditions (e.g., in relation to the west–east precipitation gradient) or in relation to patterns of human land use. Estimated genetic diversity values were erratic among fragmented populations, and our hypotheses that allelic loss through genetic drift, increase of homozygosity through high inbreeding, or strong sub-population differentiation through restricted gene flow in marginal populations could not be confirmed.

Unexpectedly, our research revealed that a degree of inbreeding is occurring in the western populations that are located in denser, wetter and less disturbed parts of the araucaria range. A possible explanation could be the negative effects of heavy rainfall on pollen movement in years with high precipitation. Furthermore, and unexpectedly, gene flow was found to be higher in the more fragmented eastern populations, which we anticipated was due to the effects of the prevailing west–east winds and resulting pollen movement.

The high within-population variation found in our study highlights the need to consider this variation in the design and implementation of restoration measures, especially in the eroded and fragmented eastern populations. Our research also showed that greater genetic variation is found in seeds during years of higher seed production and this finding has very important implication in the planning of seed harvesting for forest restoration and tree breeding purposes.

Finally, our research showed that questions of sustainability of forest management practices must be answered on a large scale. It is also important to look at both direct and indirect effects of human activities on biological processes and, more particularly, on the genetic dynamics that characterize the macro- and micro-evolutionary processes in araucaria forests. Solutions leading to forest use sustainability must be found by taking into account the broad range of economic activities of local forest users and the development options for improving management of other resources (crops, livestock, etc.) that contribute to the livelihood of local communities.

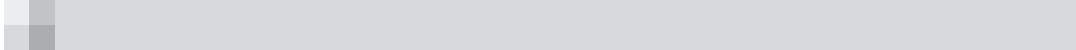
Furthermore, any potential solutions for improving management and use of pasturelands and araucaria forests in the areas surrounding the Mapuche communities studied must be discussed in wide consultation, engaging the indigenous communities, land managers, and government officials.

References

- Armesto, J.J., C. Villagrán, J.C. Aravena, C. Pérez, C. Smith-Ramírez and L. Hedin. 1995. Conifer forests of the Chilean coastal range. Pp. 156-170 in *Ecology of the Southern Conifers* (N.J. Enright and R.S. Hill, eds.). Melbourne University Press, Carlton, Victoria.

- Arroyo, M.T.K., M. Riveros, A. Peñaloza, L. Cavieres and A.M. Faggi. 1995. Phytogeographic relationships and regional richness patterns of the cool temperate rainforest flora of southern South America. Pp. 134-172 in *High Latitude Rain Forests of the West Coast of the Americas: Climate, Hydrology, Ecology and Conservation* (R. Lawford, P. Alaback and E.R. Fuentes, eds.). Springer-Verlag, Berlin, Germany.
- Bergmann, F. and H.H. Hattemer. 1998. Isozymes in forest genetics research. Pp. 227-238 in *Forest Genetics and Tree Breeding* (A.K. Mandal and G.L. Gibson, eds.). CBS Publishers and Distributors, New Delhi, India.
- Burns, B.R. 1991. The Regeneration Dynamics of *Araucaria araucana*. PhD Thesis, University of Colorado, Boulder, USA.
- Cheliak, W.M. and J.A. Pitel. 1984. Techniques for starch gel electrophoresis of enzymes from forest tree species. Patawawa National Forestry Institute, Canadian Forestry Service Information Report PI-X-42. Ottawa, Canada.
- Donoso, C. 1987. Variación natural en especies de *Nothofagus* en Chile. *Bosque* 8:85-97.
- Donoso, Z. 1998. Bosques templados de Chile y Argentina. Variación, Estructura y Dinámica. *Ecología Forestal*. Ed. Universitaria, Universidad de Chile, Santiago, Chile.
- Flueck, W.T., M. Franken and J.M. Smith-Flueck. 1999. Red deer, cattle and horses at high elevations in the Andean Cordillera: Habitat use and deer density. *Mastozoología Neotropical* 6(2):91-101.
- Funes, M.C. 1996. The European rabbit: Patterns of spread and resource availability along watersheds in northern Patagonia, Argentina. MSc thesis, University of Florida, Gainesville, USA.
- Gallo, L. 2003. Conservación, manejo y uso de sustentable de los recursos genéticos de la *Araucaria araucana* en Argentina, Comunidades Aucapán y Chiuquilihuin, San Carlos de Bariloche. Report on the project Conservation, Management and Sustainable Use of Forest Genetic Resources with Reference to Brazil and Argentina. International Plant Genetic Resources Institute, Rome, Italy.
- Gregorius, H.-R. 1974. On the concept of genetic distance between populations based on gene frequencies. Pp. 17-26 in *Proceedings of the IUFRO Joint Meeting of Working Parties on Population and Ecological Genetics, Breeding Theory and Progeny Testing*. Department of Forest Genetics, Royal College of Forestry, Stockholm, Sweden.
- Gregorius, H.-R. 1980. The probability of losing an allele when diploided genotypes are sampled. *Biometrics* 36:643-652.
- Gregorius, H.-R. and J.H. Roberds. 1986. Measurement of genetical differentiation among subpopulations. *Theor. Appl. Genet.* 71:826-834.
- Hamrick, J.L., M.J.W. Godt and S.L. Sherman-Broyles. 1992. Factors influencing levels of genetic diversity in woody plant species. *New Forests* 6:95-124.
- Hartl, D.L. and A.G. Clark. 1997. *Principles of Population Genetics* (3rd edn). Sinauer Associates, Inc. Publishers. Sunderland, USA.
- Holechek, J. and B. Gross. 1982. Evaluation of different calculation procedures for microhistological analysis. *J. Range Manage.* 35(6):721-723.
- Marchelli, P. and L.A. Gallo. 1999. Annual and geographic variation in seed traits of Argentinean populations of southern beech *Nothofagus nervosa* (Phil.) Dim. et Mil. *For. Ecol. Manage.* 121:239-250.
- Marchelli, P. and L.A. Gallo. 2001. Genetic diversity and differentiation in a southern beech subjected to introgressive hybridization. *Heredity* 87:284-293.
- Mateucci, S. and A. Colma. 1982. Metodología para el estudio de la vegetación. Secretaría General de la organización de los Estados Americanos. Programa Regional de Desarrollo Científico y Tecnológico, Washington, DC, USA.
- Montaldo, P. 1974. La bio-ecología de *Araucaria araucana* (Mol.) Koch. Instituto Forestal Latino-Americano. Boletín Técnico, 46. Universidad de Chile, Santiago, Chile.
- Nei, M. 1973. Analysis of gene diversity in subdivided populations. *Proc. Nat. Acad. Sci. USA* 70:3321-3323.
- Pastorino, M.J., L.A. Gallo and H.H. Hattemer. (In press). Genetic populations in natural populations of *Austrocedrus chilensis*, a cypress of the Andean-Patagonian Forest. *Biochem. Syst. Ecol.*

- Premoli, A. 1997. Genetic variation of two geographically restricted and widespread species of South American *Nothofagus*. J. Biogeogr. 24:883-892.
- Rechene, C. 2000. Los bosques de *Araucaria araucana* en Argentina. Estudios silvícolas. Centro de Investigación y Extensión Forestal Andino Patagónico, Esquel, Chubut, Argentina. Lehrstuhl für Waldbau und Forsteinrichtung, Universidad Técnica de Munich, Freising, Alemania.
- Sanguinetti, J., L. Maresca, M. Gonzalez Peñalba and L. Chauchard. 2000. Producción Bruta del Piñon de la Araucaria (*Araucaria araucana*) en Rucachoroi y Tromen (Parque Nacional Lanín-Argentina) años 2000-2001. Informe Interno - Administración de Parques Nacionales, Argentina.
- Sanguinetti, J., L. Maresca, M. Gonzalez Peñalba, L. Chauchard and L. Lozano. 2002. Producción Bruta de semillas de *Araucaria araucana*. 3er Informe Interno. Administración de Parques Nacionales, Argentina.
- Stoddart, L.A., A.D. Smith and T.W. Box. 1975. Range management. McGraw-Hill, New York, USA.
- Sutherland, W.J. (ed.). 1996. Ecological Census Techniques. A Handbook. Cambridge University Press, Cambridge, UK.
- Tomlinson, P.B. 2002. Crown structure in Araucariaceae. International Araucariaceae Symposium in Auckland, New Zealand.
- Veblen, T.T. 1982. Regeneration patterns in *Araucaria araucana* forest in Chile. J. Biogeogr. 9:11-28.
- Veblen, T.T. and R. Delmastro. 1976. Los recursos genéticos de *Araucaria araucana* en Chile. Información sobre Recursos Genéticos Forestales (Food and Agriculture Organization) 5:2-5.
- Veblen, T.T., T. Kitzberger, R. Villalba and J. Donnegan. 1999. Fire History in northern Patagonia: The roles of humans and climatic variation. Ecol. Monogr. 69(1):47-67.
- Wright, S. 1951. The genetical structure of populations. Ann. Eugen. 15:323-354.
- Wright, S. 1978. Evolution and genetic populations. Vol. 4 Variability within and among natural populations. The University of Chicago Press, Chicago, USA.



Chapter 7

Conservation, management and sustainable use of *Araucaria angustifolia* genetic resources in Brazil

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1. Introduction to the case study

This chapter assesses the conservation status of another araucaria species, *Araucaria angustifolia*, in the state of Paraná, Brazil. The project mainly investigated the repercussions of policy frameworks regulating access to and use of forest genetic resources of *A. angustifolia*. Mixed *A. angustifolia* forest is one of the most important biomes that occur naturally in south and southeastern Brazil. It extends from latitudes 19°30'S to 31°30'S and from longitudes 41°30'W to 54°30'W and includes a small area in the Province of Misiones in Argentina. The original *A. angustifolia* forest covered an estimated area of 200 000 km² (Maack 1950; Maack 1981), but the great value of its wood has led to a dramatic reduction in the number and size of its populations in southern Brazil. The species is today classified as vulnerable in the World Conservation Union (IUCN) 2000 Red List of Threatened Species (Hilton-Taylor 2000). The greatest extension of *A. angustifolia* in Brazil occurred in Paraná State where it originally occupied 7 378 000 hectares (Maack 1981), corresponding to 37% of the total area of that state.

2. *A. angustifolia* forest in Brazil and its exploitation

A. angustifolia (Bertoloni) O. Kuntze is a subtropical species in the Araucariaceae family, in the order Coniferales. It is known in Brazil as araucaria, parana-pine, pinheiro-do-paraná or pinheiro-brasileiro. Trees can reach 35 to 60 m in height and 0.8 to 2 m in dbh (diameter at breast height) (Klein 1960). Araucaria is usually dioecious, rarely monoecious, with 2n (diploid chromosome number) = 26 (Bandel and Gurgel 1967). Like other conifers, araucaria is wind pollinated, and pollen maturation and pollination in Brazil occur from August through October. The seed cones begin to mature two years after pollination, and the complete cycle from primordial carpel to seed takes four years (Shimoya 1962). Young trees begin to set seed between 12 and 15 years of age. Seeds are dispersed from May through August (Carvalho 1994).

Primary araucaria forest in Brazil is composed of mixed species and is usually characterized by having three canopy layers. The uppermost canopy consists of crowns of the older araucaria trees that allow penetration of a considerable amount of light. The middle

canopy is mostly formed by Lauraceae species and the lowermost canopy by species of Myrtaceae and by *Ilex paraguariensis* (Aquifoliaceae), called erva-mate.

On the basis of photosynthetic and growth responses of araucaria trees grown under different light conditions (Inoue *et al.* 1979; Einig *et al.* 1999), *A. angustifolia* was found to be well adapted to moderate shade. However, since there is little information on the species' behaviour under natural light conditions, the observed absence of seedlings in some shaded environments in the wild led to the commonly held belief that araucaria was a sun-loving, pioneer species that would not regenerate in forest understorey.

The commercial exploitation of araucaria wood was one of the most important economic activities in southern Brazil until the end of the 1970s (Guerra *et al.* 2002). Araucaria wood was a major resource for social and economic development in this region, providing high-quality timber for construction and furniture, and long-fibre wood for pulp and paper industries (Carvalho 1994). The araucaria tree also supplies several nonwood forest products (NWFPs): buds and knurs (knots in the tree trunk) are used in folk medicine (Marquesini 1995) and handicrafts, and seeds are consumed as food by humans and livestock for their high nutritional value (Carvalho 1994) (Figure 1).

Exploitation of araucaria forest increased after 1934 when road construction projects started and forest industries were established in Paraná and Santa Catarina states. This period also coincided with a growth in demand for agricultural and pasture lands, all of which was driven by population growth in Brazil. More than 15 million m³ of araucaria timber was exported between 1958 and 1987, and this wood was the most important Brazilian forest product until the 1970s (Reitz and Klein 1966, cited by Guerra *et al.* 2002). By 1978, araucaria forest area had been reduced to 8% of its original extent (FUPEF 1978). Currently less than 3% of the original araucaria forest cover remains, with only 0.7% occurring in the form of primary forest (FUPEF 2001).

According to Laboriau and Matos Filho (1948, cited by Guerra *et al.* 2002), most of the araucaria forest lands were exploited in three phases: first, the most commercially valuable trees were harvested; second, among the trees left after phase one, those trees with high-quality wood were logged; and third, the lands were burned to create pasture and agricultural areas.



Figure 1. Indigenous people from Manguairinha (Paraná) preparing traditional handicrafts with nonwood forest products (photo: J.V. Bittencourt).

Since the start of commercial exploitation, the practices adopted to harvest araucaria have not been inspired by a philosophy of sustainable use, and they resulted in a dramatic change in the appearance of araucaria forest resources. Surveys of secondary araucaria forests harvested more than 50 years ago, carried out within this project, showed that the numbers of juvenile and adult araucaria trees that were expected to have originated from natural regeneration were not found. Only when weeds were controlled and when the light needed for regeneration was manipulated by silvicultural treatments did new trees do well.

In 1993 the Brazilian government introduced new legislation (*Decreto-lei* no 750) that established rules for sustainable management of araucaria forests. According to the rules adopted in Paraná State, only trees with more than 40 cm dbh could be harvested, and at least ten mother-trees in this diameter class per hectare had to be left unharvested. Araucaria harvesting was prohibited by law in 2001 when surveys showed less than 3% of the original araucaria forest cover remained as fragmented forest land (FUPEF 2001).

The transformation of most araucaria native forests into pastures and agricultural lands caused the extirpation of many natural populations. Before the adoption of sustainable management regimes, the dysgenic selection that had occurred in exploited populations contributed to reducing their genetic variability to levels now low enough to compromise their use for conservation and breeding purposes. As a result, conservation and breeding programmes are urgently needed that are based on an understanding of the genetic structures of the remaining populations.

Awareness is growing in Brazil that *A. angustifolia* is vulnerable to genetic erosion. While not yet under threat of extinction, several geographic populations have already been lost. Many areas once occupied by araucaria are now used for cattle grazing or for plantations of fast-growing exotic trees. Reforestation programmes have to date been limited both in number and in success, and this may be related to the limited information on many important ecological and physiological features of the species.

The Brazilian government is now promoting several initiatives to protect araucaria genetic resources. For example, the harvesting of naturally regenerated araucaria trees is now prohibited by law (PN-COMANA – National Programme of the National Environmental Council – 278, 18 July 2001). And more recently, the federal government selected five regions in Paraná State where genetic conservation of araucaria forests is now a priority (PN MMA – National programme of the Ministry of the Environment – 507, 20 December 2002). Under this legislation, araucaria or other native species must be used in any expansion of forest area plantations. However, financial constraints have limited the Brazilian government's ability to buy properties in order to establish conservation units like national parks or reserves.

The government of Paraná State has been involved in this effort through the establishment of 'biodiversity corridors' by state agencies. This is a multi-institutional initiative that aims to increase connectivity between the araucaria forest fragments (Figure 2, overleaf) that are of interest to genetic conservation. This project is supported by the Critical Ecosystem Partnership Fund (CEPF), Conservation International (CI), the Global Environment Facility (GEF), the Japanese government, the MacArthur Foundation and the World Bank. The project is part of a programme designed to safeguard the world's threatened biodiversity hotspots in developing countries.

The present poor condition of the natural araucaria forests in Brazil is a consequence of the fact that there is no adequate management model for native forests that combines ecological, genetic, social and economic factors. Because araucaria plantations are less profitable than other land-use options like pine plantations or agricultural crops, this situation perhaps results from the typical way that most Brazilians view forests: that is, simply as a source of timber.

Local experience has led us to conclude that there are two viable approaches to achieving conservation of araucaria genetic resources. Both options will require the scientific knowledge obtained in studying the genetic diversity found in the remaining araucaria populations:



Figure 2. Remnants of *A. angustifolia* forest stands (photo: J.V. Bittencourt).

(i) Establishing new araucaria plantations by farmers and private forestry companies in residual araucaria forests. This option will require genetically improved seeds and efficient agrosilvicultural techniques to increase the profitability of the araucaria plantations.

(ii) Government or private initiatives establishing *in situ* and *ex situ* conservation programmes for araucaria. These initiatives will require substantial financial resources in order to purchase, establish and maintain conservation areas. The creation of private nature protected reserves (RPPN) in araucaria forest areas, mainly by volunteer landowners, could lead to successful outcomes. But local stakeholders would like to see responsibility for this undertaking shared with the government through incentives and subsidies. Local landowners have expressed the opinion that the government should either buy the land to establish conservation areas, or it should provide subsidies to local stakeholders to maintain the current araucaria forest cover intact until scientifically based, sustainable forest management practices are defined and adopted. Because this position has led to delays, several araucaria forest fragments, extremely valuable for the conservation of the species genetic resources, have yet to be protected.

There is yet another management approach for araucaria forests that we feel should be considered. It is based on the traditional community-based system for natural resource management used by small-scale farmers in the Paraná forest, called 'faxinal' (Box 1). Historically, the Paraná countryside was organized into communities that kept agricultural land for individual use and forest land for common use. The system is still practised in some parts of Paraná State. Nevertheless, while many of the existing araucaria forest fragments are found within these communal land management units, the system is not now sustainable for the species in the long run because the forest is being used for grazing, and this adversely affects natural regeneration of araucaria trees.

With modifications to the traditional faxinal system, we feel that the long-term conservation and sustainable use of araucaria would stand a good chance of success.

Box 1. Faxinal: a traditional agroforestry system in the Brazilian *A. angustifolia* forest region

Some rural communities in Paraná State still practise a traditional agroforestry system called faxinal, introduced by Ukrainian immigrants in the 19th century. In this system, the community uses forest land for timber, NWFPs, and for grazing domesticated animals (mainly cattle, horses and pigs). The system is designed around collective use of land for animal production and is associated with low extraction of forest products, resulting in a regular flow of income into the communities. Nonwood tree species like *Ilex paraguariensis*, whose leaves are used to make a tea called erva-mate, play an important role in the system. *I. paraguariensis* occurs naturally in araucaria forests and its survival is dependent on their conservation.

Individual families in faxinal communities cultivate annual crops such as maize, black beans and cassava outside the native forest land, which is surrounded by a fence in order to keep the animals inside and away from the cultivated areas. Some medicinal plants are grown in home gardens.

The faxinal system has contributed to the conservation of many of the remaining araucaria forests. Despite their ecological, social and cultural advantages, there are fewer and fewer faxinais today, mostly because traditional practices make it hard for them to compete with the 'higher-tech' farming systems found in neighbouring areas.

While the faxinal is a subsistence system with much lower agricultural and livestock production than modern farms, improvements in communal natural resource management could help the smallholder families remain on their traditional lands, improve their living conditions, and at the same time secure the conservation of the natural araucaria forest remnants.

3. Agricultural properties in Paraná State

The State of Paraná covers 19.5 million hectares, representing 2.3% of the country of Brazil. Of its population of 9.5 million people, 82% reside in urban areas, while 18% live in the countryside. Paraná State's agricultural production is the highest in Brazil, representing 23% of agricultural production in the country. Around 370 000 rural properties are spread across the state territory. Table 1 provides a synthetic picture of private property sizes and their distribution in the State (Turra 2003).

3.1 *A. angustifolia* forest types and threats

The natural range of araucaria forest in Brazil lies within the southern part of the country. The natural vegetation pattern of the araucaria region is a matrix of patches of subtropical rainforest interrupted by often-extensive grasslands (Klein 1963). *A. angustifolia* (Bert.) O. Ktze represents more than 40% of the trees in this forest type (Longhi 1980; Oliveira

Table 1. Size of properties and their relative percentages in Paraná State.

Size (ha)	Percentage of all properties	Percentage of agricultural land
< 10	42	5
10 to 50	44	23
51 to 100	7	11
> 100	7	61

Source: Turra 2003.

and Rotta 1982). It is the only species in its genus in Brazil and, along with *Podocarpus lambertii* Klotz (pinheiro-bravo) and *Podocarpus sellowii* (pinheiro-bravo-da-folha-larga), is one of only three native conifers to occur in Brazil.

In 2002, in conjunction with this BMZ/IPGRI/FUPEF (German Federal Ministry for Economic Cooperation and Development/International Plant Genetic Resources Institute/ Foundation for Forest Research in Paraná) subproject, other research activities on araucaria were being implemented within the framework of the FUPEF/PROBIO (National Biodiversity Project) project that focused on research and development of demonstration projects and assessments for the conservation and sustainable use of biodiversity in Brazilian biomes. The research carried out within the PROBIO framework helped to locate remaining fragments of araucaria forests in Paraná State (Figure 3), and to select sample plots for the BMZ/IPGRI/FUPEF subproject. Forest fragments were identified using satellite images (geographic information system – Landsat 1998/99). A field survey was conducted at 304 randomly selected points in the different araucaria forest ecological regions in order to map the distribution of fragments and to evaluate their phytosociological characteristics. A rapid ecological survey was undertaken using soil and vegetation characteristics to detect occurrences of human disturbance, and to check the quality of these remnants.

The FUPEF/PROBIO research group identified 66 109 ha of primary araucaria forest (0.8% of the original area in Paraná State), 1.2 million ha (14.5%) of secondary forest and 1.2 million ha (14%) of forest in an early regeneration phase (PROBIO/FUPEF 2002). Forests dominated by *A. angustifolia* covered 75 783 ha (or 0.91% of Paraná State), and the group produced 53 maps (1:100 000) of the araucaria forest bioregion in Paraná State. Natural araucaria distribution was found to be 12.4% greater than the area previously determined by Maack (1981). Nevertheless, the araucaria forest is highly fragmented, with most fragments concentrated in the south-central part of the state. This is a mountainous area now dominated by agricultural land (60%), pastureland (18%) and forest (4%). It is comprised today of 112 000 small- to medium-sized properties, 86% of which are less than 100 ha. Historically, most of these properties were organized as faxinal agroforestry systems.

Using non-adaptive markers, several researchers have identified genetic variation across the species' natural range. For example, Reitz and Klein (1966) described



Figure 3. Fragments of araucaria forest in Paraná State, Brazil. The smaller map shows the boundaries of Paraná State and, within it, in dark grey, the range of *A. angustifolia*, enlarged in the bigger map. In the small map, white indicates areas without forest cover. In the large map, black dots indicate the distribution of araucaria populations (adapted from PROBIO/FUPEF 2002).

nine botanical varieties of *A. angustifolia* based on ripening time and seed colour, and an additional variety was identified later by Mattos (1994). Reitz and Klein (1966) and Kageyama and Jacob (1980) detected genetic variation within and among three natural populations, and *A. angustifolia* collected from five Brazilian states showed statistically significant differences in wood production when measuring quantitative traits (Monteiro and Speltz 1980). Studies of araucaria using biochemical and molecular markers have attempted to determine the species' genetic diversity across its natural range (Mazza 1997; Schlögl 2000; Shimizu *et al.* 2000). It was found that lower genetic similarities among araucaria populations were associated with larger geographical distances.

3.2 Forest genetic conservation

The need for genetic conservation of araucaria forests has been recognized in Brazil since early in the last century. But few government initiatives were undertaken to establish conservation areas (such as National Parks or Reserves) in the natural habitat of araucaria, though some plantation field trials occurred between the 1950s and the 1980s. Until 1979, only about 90 000 ha of plantations had been established in Brazil (Shimizu and Oliveira 1980). The reasons for this limited interest in araucaria plantations are thought to be: (i) lack of knowledge about the most suitable abiotic factors (soil and climate) for wood production; (ii) difficulties in obtaining seeds from selected sites; (iii) the nonavailability of genetically improved seeds; (iv) the absence of specific silvicultural techniques; and (v) the slow growth rate of the species when compared to *Pinus* or *Eucalyptus* species (Shimizu and Oliveira 1980).

Araucaria forest management procedures adopted in Brazil during the 19th and 20th centuries allowed for the removal of trees with 40 cm dbh or more. It was believed that the remnant trees would retain the variability of the population's gene pool and would produce the seeds needed for natural regeneration. However, evaluation of this management approach by IBAMA (Brazilian Institute of Environment and Natural Resources) showed that it produced rates of natural regeneration lower than expected. As a result, the legislation was changed in 1990 (O.S. IBAMA-PR 024/90) whereby at least 35% of trees above 40 cm dbh were required to be left uncut. But fragments of araucaria forest continued to be harvested until 2001 when a new law (PN CONAMA 278) was passed that prohibited harvesting araucaria trees from any natural population in Brazil. This law brought significant changes to araucaria forest management, with NWFPs becoming the main alternative cash-generating option for araucaria forests. Extraction of NWFPs required a different approach to forest management, with the participation of local people being seen as crucial to establishing viable and useful management strategies.

4. Research

4.1 Objectives and methods

A preliminary assessment of the ecological characteristics of fragmented araucaria forests was carried out. We felt that data from this assessment would be very important to the implementation of any *in situ* conservation strategies for forest genetic resources (FGRs) that were subsequently adopted. Representative 20 x 20 m plots were established in four fragments subjected to different management regimes and evaluated. Plots were located at Campina da Alegria in Santa Catarina State (latitude 26°52'07 S and longitude 52°08'01 W, altitude 880 m asl).

An evaluation of the genetic diversity within and between several key remaining populations of araucaria in forest fragments was also undertaken in order to obtain data that we hoped to use in planning and implementing genetic conservation strategies for the species. Sample sites in Paraná State were chosen at Turvo, Cândói, Mangueirinha and Palmas. Each study site was characterized by a different forestry management practice (Figure 4 and Table 2, both overleaf). RAPDs (random amplified polymorphic DNA) markers were used to evaluate genetic diversity within and between araucaria populations.



Figure 4. Location of Turvo (1), Candói (2), Mangueirinha (3) and Palmas (4) municipalities in Paraná State, Brazil.

Table 2. Forest type, altitude, and location of the study site.

Area	Forest type*	Municipality	Altitude (m asl)	Latitude	Longitude
1	A	Turvo	1 040	25° 02' 34" S	51° 31' 47" W
2	B	Candói	950	25° 34' 20" S	52° 03' 21" W
3	C	Mangueirinha	921	25° 56' 28" S	52° 10' 32" W
4	A	Palmas	1 035	26° 29' 03" S	51° 59' 26" W

*A = *A. angustifolia* and grassland or pioneering plant formations. B = *A. angustifolia* and *Ocotea* spp. association. C = *A. angustifolia* and *Sloanea lasiocoma* K. Schum. transition area between araucaria and riparian forest.

The legislative policy framework was also investigated in terms of its potential repercussions on the status of araucaria FGRs. The law introduced in 1990 required 35% of araucaria trees above 40 cm dbh to be left in place, and we believed it was likely that this change affected the genetic structure of the population of araucaria stands. In order to assess the effects of this law, we compared genetic variation in three populations of araucaria at Campina da Alegria. The first (= pop 1) was a natural population with little human disturbance, the second (= pop 2), a managed population, and the third (= pop 3) was a newly established field progeny trial grown from seeds collected from harvested trees and planted 20 years ago. Seeds for the field progeny test were collected at a sampling intensity of one parent tree for each 2.4 ha of forest. Twenty-nine RAPD primers were selected to study the genetic distance among these three populations.

A fourth research activity focused on identifying molecular markers that could be used to determine the sex of juvenile araucaria plants. Because the proportion of male and female trees in a natural population is 1:1 (Bandel and Gurgel 1967), for seed production purposes we felt it would be useful to be able to determine the sex of seedlings in order to guide small farmers in their restoration plans, and because we felt that regeneration of the species was likely to benefit from the maintenance of a proper ratio between female and male individuals. A bulk of male and female trees was assembled and a DNA analysis (using primers from Operon Tech, Alameda, California, USA) was undertaken to search for sex-specific markers by extracting DNA from a sample of ten female (Figure 5) and ten male plants.



Figure 5. Female cone of *A. angustifolia* (photo: J.V. Bittencourt).

Because of araucaria's economic importance, several forest research institutions in Brazil began work on the species in the 1950s, though there was little follow-up on the population trials that were started (FUPEF 1978). One of the objectives of our BMZ-funded subproject was to retrieve information on araucaria growth performance from those early *ex situ* field trials that were established with seeds from different sites and progenies.

In the area of social science research, we carried out a survey of the traditional use of forest resources by a local community that had previously been organized as a faxinal. We gathered information on the knowledge associated with the management and use of araucaria forests by local people (Figure 6, overleaf), and we attempted to assess local perceptions of conservation issues. The community that we studied was chosen for its existing knowledge of forest products that were still important in making a living. Principal sources of income within the community were agriculture, livestock production and, more recently, medicinal plant gathering from araucaria forests. Data were thus collected on current uses of araucaria forest products and information was obtained both on local people's feelings about araucaria conservation and on the effects of the 2001 araucaria logging ban on local people's livelihoods.

4.2 Results

Table 3 (on p. 143) summarizes the ecological characteristics of the secondary araucaria forest plots at Campina da Alegria. The majority of young araucaria trees (individuals with a height of less than 3 m) were found growing under hardwood tree species mainly of the Lauraceae family, but a few were found growing under mature araucaria trees, apparently indicating better regeneration of araucaria under the canopy of other species. No seedlings were found in either deep shade or bright light, seemingly indicating that light is an important factor for natural araucaria regeneration and growth at a young age. We also observed that many of the young trees growing under hardwood crowns had low stem and crown quality. We concluded that further studies were needed in order to determine a suitable light management regime to take araucaria seedlings through to adulthood. This was important because it stimulated our preliminary thinking on the light conditions that would be needed to establish araucaria seedlings in plantations and recovery programmes.



Figure 6. Member of an indigenous community living inside the reserve of Mangueirinha (Paraná), holding a female cone of *A. angustifolia* (photo: J.V. Bittencourt).

The level of genetic variability found at the four sites investigated in Paraná State (Turvo, Cândói, Mangueirinha and Palmas) was 15% between populations and 85% within populations. Turvo and Mangueirinha populations displayed a larger percentage of polymorphic loci (81.97% and 77.05%, respectively), followed by Palmas (74.29%) and Cândói (68.80%) populations. Based on these results, *in situ* conservation strategies should prioritize the Turvo and Mangueirinha populations.

In assessing the impact of pre-1990 management on FGRs in three populations of araucaria from Campina da Alegria, we found genetic segregation among the three populations that could be linked in some way to human intervention (Figure 7, opposite). When comparing the natural (pop 1) and managed populations (pop 2), a reduction of 11.58% in the polymorphic loci was found. The difference increased to 27.43% when the natural population (pop 1) was compared to the progeny trial population (pop 3), showing a clear segregation of the progeny trial from both the natural and the managed populations.

Based on these results, it would seem that the management regime adopted in the 1990s (leaving 35% mature araucaria trees) was sustainable from a genetic point of view, as it did not significantly reduce the genetic variability of the new populations established after logging in the secondary forest. However, these findings, in order to be conclusive and useful in developing guidelines for *in situ* genetic conservation programmes, would need to be supported by a more comprehensive study based on a larger sample of parent trees. Ideally, the collection of plant material for *ex situ* conservation studies should be done by harvesting seeds from a sample larger than one mother tree every 2.4 hectares or every 50 trees.

Table 3. Evaluation of araucaria trees in the four plots of secondary forest at Campina da Alegria in the State of Santa Catarina, Brazil.

Evaluation	Plot 1	Plot 2	Plot 3	Plot 4
Number of araucaria trees	4	15	5	6
Crown shape of araucaria trees	Concave, plane	Convex, plane (7 young trees)	Convex	Convex
Basal area (m ²) of all araucaria trees in the plot	28.8	14.1	7.2	14.2
Number of hardwood species	9	16	10	12
Number of individuals of hardwood species	21	57	28	27
Total basal area (m ²) of all hardwood species in the plot	40.2	69.5	9.5	3.5
Spatial density of <i>Bambusa</i> sp. shoots, which compete with araucaria regeneration	Very high	Rare	Rare	High
Individuals of <i>Dicksonia</i> sp. per plot, which compete with araucaria regeneration	8	-	18	-
Density of herbaceous species cover	Low	Low	High (Pteridophyte)	High
Light conditions (shade) Visual observation	Very shady	Little shade	Highly shady	Highly shady
Presence of <i>Ilex paraguariensis</i>	yes	-	yes	yes
Presence of <i>Ocotea porosa</i>	yes	-	-	yes

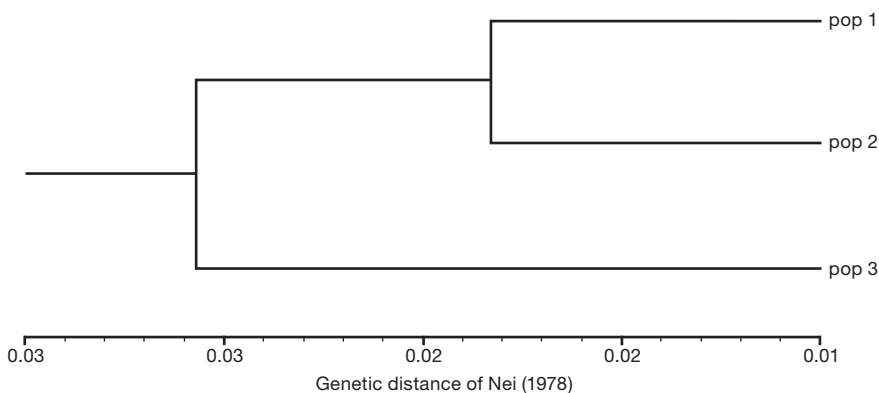


Figure 7. Dendrogram representing the genetic distance (Nei 1978). Pop 1 = natural population with little human intervention; pop 2 = a secondary, managed population; pop 3 = field progeny test established with seeds collected from harvested trees (adapted from Medri *et al.* 2003).

The screening that we undertook to identify polymorphic bands for recognizing sex differences was carried out using 600 RAPD primers. The molecular markers identified turned out not to be linked to sex determination genes, therefore the proposed objective could not be achieved within the project's time frame.

Table 4 summarizes the information we assembled on araucaria field trials and gene banks in Brazil. Some of these trials involved natural population growth tests and others combined these with progeny growth trials. This initiative generated data on the genetic composition of different natural plant populations from different regions of Brazil in the past, and as such will be very useful if and when recovery programmes are implemented.

Results for only a few of these field trials were published. For example, Shimizu and Higa (1980) published the results of a 6-year field trial on plants from 12 natural populations established at Itapeva, São Paulo. The trials revealed the existence of significant genetic variation among the regions, and that populations from the same region were more similar to each other than they were to populations from distant regions. These findings support the hypothesis of Gurgel and Gurgel Filho (1973) on the existence of ecotypes or geographical races of *A. angustifolia*, although individual trees grown from seeds gathered nearer to the planted areas did not show higher survival or growth rates. In general, populations from rainier southern Brazil showed the best growth, and those from the drier northern region the poorest growth, with a few exceptions. However, a strong correlation was found between plant height and latitude, with individuals from a few northern provenances tending to grow even better in Itapeva, in the south, than individuals from southern populations (Table 5, on p. 146), indicating a considerable plasticity in this species' growth pattern.

While work like that of Shimizu and Higa (1980) is promising, much more similar work needs to be done on araucaria in order to capitalize on these early field trials. Such information would be most useful in helping us develop breeding strategies for different parts of the country.

While investigating the socioeconomic issues, we found that because current legislation limits araucaria logging, local people have shifted their economic focus in araucaria forests to NWFPs. Within selected households, collecting araucaria seeds for food or for seedling production was found to be an excellent economic alternative to logging, even though the seed production period is limited to the months of April, May and June. One hectare of araucaria forest has been estimated to produce an average of 180 kg of seeds annually. Using market prices of US\$0.60 per kg, this equates to a gross annual income of US\$108/ha for araucaria seed alone.

While this remains a modest income-generating activity, the community survey also revealed that local farmers used more than 70 other species of shrubs and herbaceous plants solely for medicinal purposes (Bittencourt and Higa 2004). While our project focused on traditional use of medicinal plants, local people in their day-to-day lives used many other NWFPs. The maintenance of healthy *A. angustifolia* forest ecosystems thus provides many other benefits that are critical to local community well being. Though these benefits are hard to quantify in monetary terms, they should not be discounted in determining the conservation value of araucaria forest genetic resources.

5. Conclusions and recommendations

This study revealed that the current forest legislation in Brazil is inadequate to secure the maintenance of *A. angustifolia* forests. Indeed, the 2001 law seems to be having a negative effect on forest cover. Many local people are no longer planting the species because they are worried that their trees will become untouchable owing to the current logging ban.

The research carried out within the context of the FUEPF/PROBIO project (2001) confirmed earlier evidence of pronounced geographic differentiation of the species from a genetic aspect. Regions with greater genetic diversity were identified, and these should become priority areas for conservation. In particular, among the populations studied in our research, two (Turvo and Mangueirinha) were found to be most suitable for *in situ* genetic conservation programmes in Paraná State.

Table 4. Field trials with *A. angustifolia* in Brazil.

Institution *	Planting year	Structure	Number of regions	Location of the field trials **
IF SP	1952	Population	4	Santa Rita do Passa Quatro, SP
IF SP	1967	Population	13	Batatais, SP
IF SP	1967	Population	17	Campos do Jordão, SP
IF SP	1967	Population	19	Avaré, SP
IF SP	1967	Population	15	São Miguel Arcanjo, SP
Klabin	1967	Population	24	Telêmaco Borba, PR
IBAMA	1967	Population	23	Irati, PR
Swedish Match	1967	Population	11	São João do Triunfo, PR
Rigesa	1967	Population	23	Três Barras, SC
IBAMA	1974	Population	18	Três Barras, SC
Guaraci	1974	Population	18	Guaraci, PR
Sguario	1974	Population	18	Itapeva, SP
IF SP	1974	Pop/progeny	5 Populations	Itapeva, SP
IF SP	1974	Pop/progeny	5 Populations	Itapetininga, SP
IBAMA	1975	Pop/ progeny	3 Populations	Três Barras, SC
IPEF	1975	Pop/ progeny	3 Populations	Guarapuava, PR
IPEF	1975	Pop/ progeny	3 Populations	Lages, SC
IPEF	1975	Pop/ progeny	3 Populations	Campos do Jordão, SP
IF SP	1980	Pop/ progeny	12	Itapeva, SP
IF SP	1980	Pop/ progeny	12	Itapetininga, SP
EMBRAPA	1980	Pop/ progeny	12	Colombo, PR
UFPR	1980	Pop/ progeny	12	Quatro Barras, PR

* Institutions: IF SP = São Paulo State Forest Institute; Klabin = Klabin do Paraná Forest Industries; IBAMA = Brazilian Agency for the Environment; Rigesa = Mead Westvaco Rigesa Forest Industries; Swedish Match, Guaraci, Sguario = forest industries; IPEF = Brazilian Institute of Forest Research; EMBRAPA = Brazilian Corporation for Agricultural and Livestock Research; UFPR = Federal University of Paraná.

** Locations: SP = São Paulo State; PR = Paraná State, SC = Santa Catarina State.

Table 5. Araucaria plants from different populations grown at Itapeva, Brazil.

Population	Latitude	Altitude (m)	Rainfall (mm)	Distance from origin (km)
01. Quedas do Iguaçu, PR	25°30'	650	1 500	380
02. Cascavel, PR	25°02'	750	1 662	425
03. Irati, PR	25°30'	880	1 442	210
04. Itapeva, SP	24°24'	900	1 400	0
05. Itatiaia, RJ	22°23'	2 100	2 416	490
06. S. Fco. De Paula, RS	29°20'	910	2 252	575
07. Telêmaco Borba, PR	24°17'	900	1 421	185
08. Barracão, PR	26°13'	835	1 686	500
09. Chapecó, SC	27°07'	675	2 180	475
10. Caçador, SC.	26°47'	960	1 567	325
11. Santa Maria, RS	29°20'	450	1 767	765
12. Cocaina*, SP	22°50'	1 400	1 681	480
13. São Joaquim, RS	28°19'	1380	1 593	450
14. Campo Mourão, PR	24°23'	800	1 640	345
15. Campos do Jordão, SP	22°44'	1 630	1 350	395
16. V. Grande do Sul*, SP	21°50'	1 000	1 200	355
17. Passo Fundo, RS	28°15'	708	1 650	550
18. Lauro Muller, SC	28°26'	250	1 438	465

Distance from origin (km) = distance between sites where populations originate and the site where seed from these populations was planted (Itapeva). * = seeds collected from plantations. SP = São Paulo State; PR = Paraná, SC = Santa Catarina; RS = Rio Grande do Sul; RJ = Rio de Janeiro. Source: Shimizu and Higa (1980).

We recommend that *in situ* conservation actions and seed collecting for *ex situ* programmes and restoration actions be implemented as a high priority within the remaining 66 000 ha of mature araucaria forest and within the 76 000 ha of mixed araucaria forest in Paraná State.

Sustainable management practices at the more degraded sites should be adopted, as araucaria responds well to silviculture and more is becoming known of its flexible reaction to different light conditions. Management of araucaria should be concentrated within the current 2.4 million ha of secondary araucaria forest, and should include augmentation

not only of *A. angustifolia*, but also of *Ocotea porosa* and *Ilex paraguairiensis*. Control of competitors like *Bambusa* sp. and *Dicksonia* sp. should also be included. Biodiversity corridors should be established to link the existing araucaria forest fragments located in Paraná State.

Field trials involving araucaria trees from populations adapted to variable environmental conditions were undertaken in the past but only partially completed. Further research on growth performances of the populations sampled should be carried out to make appropriate plant material available for reforestation and forest recovery purposes. Northern provenances should be preferentially considered for commercial plantations, given their seemingly better growth performance.

The present investigation was actively supported by the involvement and contribution of local communities whose livelihoods depend on the many NWFPs extracted from araucaria forests surrounding their villages and towns. Several important medicinal plants were found to grow under the araucaria canopy. Sustainable management of araucaria forest fragments by small landowners for the purpose of extracting NWFPs could be a potentially viable economic option for local people.

References

- Bandel, G. and J.A.A. Gurgel. 1967. Proporção do sexo em *Araucaria angustifolia*. Silvicultura em São Paulo 6:209-220.
- Bittencourt, J.V. and A. Higa. 2004. Final report of the IPGRI/BMZ-funded project, Conservation, Management and Sustainable Use of Forest Genetic Resources with Reference to Brazil and Argentina. International Plant Genetic Resources Institute (IPGRI), Rome, Italy.
- Carvalho, P.E.R. 1994. Espécies florestais brasileiras: recomendações silviculturais, potencialidades e uso da madeira. Empresa Brasileira de Pesquisa Agropecuária – Centro Nacional de Pesquisa de Florestas (EMBRAPA-CNPQ), Colombo, Brazil.
- Einig, W., A. Mertz and R. Hamp. 1999. Growth rate, photosynthetic activity, and leaf development of Brazil pine seedlings (*Araucaria angustifolia* [Bert.] O. Bert.). Plant Ecol. 143:23-28.
- FUPEF (Fundação de Pesquisas Florestais do Paraná). 1978. Inventário florestal do pinheiro no sul do Brasil. Relatório Final. Curitiba, Brazil.
- FUPEF (Fundação de Pesquisas Florestais do Paraná). 2001. Projeto de conservação e utilização sustentável da diversidade biológica – PROBIO, subprojeto Conservação do Bioma Floresta com Araucária. Curitiba, Brazil.
- Guerra, M.P., V. Silveira, M.S. dos Reis and L. Schneider. 2002. Exploração, manejo e conservação da araucária (*Araucaria angustifolia* [Bert]). Pp. 85-101 in Sustentável Mata Atlântica: A Exploração de Seus Recursos Florestais (L.L. Simões and C.F. Lino, eds.). Editora Senac São Paulo, Brazil.
- Gurgel, J.T.A. and O.A. Gurgel Filho. 1965. Evidências de raças geográficas no pinheiro brasileiro *Araucaria angustifolia* (Bert.) O. Ktze. Ciência e Cultura 17:33-39.
- Gurgel, J.T.A. and O.A. Gurgel Filho. 1973. Caracterização de ecótipos, em âmbito nacional para o pinheiro brasileiro *Araucaria angustifolia* (Bert.) O. Ktze. Silvicultura em São Paulo 8:127-134.
- Hilton-Taylor, C. 2000. 2000 Red List of threatened species. International Union for the Conservation of Nature (IUCN), Gland, Switzerland.
- Inoue, M.T., F. Galvão and D.V. Torres. 1979. Estudo ecofisiológico sobre *Araucaria angustifolia* (Bert.) O. Ktze: Fotossíntese em dependência da intensidade luminosa. Floresta 11:7-11.
- Kageyama, P.Y. and W.S. Jacob. 1980. Variação genética entre e dentro de populações de *Araucaria angustifolia* (Bert.) O. Ktze. Pp. 83-86 in IUFRO (International Union of Forest Research Organizations) Meeting on Forestry Problems of the Genus Araucaria, 1979, Curitiba, Fundação de Pesquisas Florestais do Paraná (FUPEF), Curitiba, Brazil.
- Klein, R.M. 1960. O aspecto dinâmico do Pinheiro do Paraná. Sellowia 12:17-44.

- Klein, R.M. 1963. Observações e considerações sobre a vegetação do Planalto nordeste catarinense. *Sellowia* 15:39-56.
- Laboriau, L.F.G. and A. Matos Filho. 1948. Notas preliminares sobre a Região da Araucária. *Anuário Brasileiro de Economia Florestal*, 1. Rio de Janeiro, Brazil.
- Longhi, S.J. 1980. A estrutura de uma floresta natural de *Araucaria angustifolia* (Bert.) O. Ktze, no sul do Brasil. MSc Thesis, Universidade Federal do Paraná, Curitiba, Brazil.
- Maack, R. 1950. Notas complementares a apresentação preliminar do mapa fitogeográfico do Estado do Paraná (Brasil). *Arquivos do Museu Paranaense*, 7:351-361.
- Maack, R. 1981. Geografia física do Estado do Paraná. Livraria José Olympio, Rio de Janeiro, Brazil.
- Marquesini, N.R. 1995. Plantas usadas como medicinais pelos índios do Paraná e Santa Catarina, sul do Brasil: guarani, kaingang, xokleng, ava-guarani, kraô e cayua. Universidade Federal do Paraná, Curitiba, Brazil.
- Mattos, J.R. 1994. O Pinheiro Brasileiro (2nd edn.), Lages, Brazil.
- Mazza, M.C.M. 1997. Use of RAPD markers in the study of genetic diversity of *Araucaria angustifolia* (Bert.) populations in Brazil. Pp 103-111 in *Recent Advances in Biotechnology for Tree Conservation and Management* (S. Bruns, S. Mantell and A.M. Tragardh Viana, eds.). International Foundation for Science, Stockholm, Sweden.
- Medri, C., P.M. Ruas, A.R. Higa, M. Murakami and C. de F. Ruas. 2003. Effects of management on the genetic variability in a population of *Araucaria angustifolia* (Bert.) O. Kuntze. *Silvae Genet.* 52(5/6):202-205.
- Monteiro, R.F.R. and R.M. Speltz. 1980. Ensaio de 24 procedências de *Araucaria angustifolia* (Bert.) O Ktze. Pp 181-200 in *International Union of Forestry Organizations (IUFRO) Meeting on Forestry Problems of the Genus Araucaria*. FUPEP (Fundação de Pesquisas Florestais do Paraná), Curitiba, Brazil.
- Nei, M. 1978. Estimation of average heterozygosity and genetic distance from a small number of individuals. *Genetics* 89:583-590.
- Oliveira, Y.M.M. and E. Rotta. 1982. Levantamento da estrutura horizontal de uma mata de araucária no primeiro planalto paranaense. *Bol. Pesq. Florestal*, Colombo 4:1-45.
- PROBIO/FUPEP (Fundação de Pesquisas Florestais do Paraná). 2002. Projeto de conservação e utilização sustentável da diversidade biológica Brasileira (PROBIO). Relatório de Atividades, período 1996-2002. Ministério do Meio Ambiente, Brasília, Brazil.
- Reitz, R. and R.M. Klein. 1966. *Araucariaceas*. Herbário Barbosa Rodrigues, Itajaí, Santa Catarina, Brazil.
- Roderjan, C.V., F. Galvão, Y.S. Kuniyoshi and G. Hatschbach. 2002. As unidades fitogeográficas do Estado do Paraná. *Ciência e ambiente*, Santa Maria, Universidade Federal de Paraná (UFPR), Brazil, 24:75-92.
- Schlögl, P.S. 2000. Análise da diversidade genética em regiões não codificadoras de DNAs de cloroplastos em *Araucaria angustifolia* por PCR-RFLP. MSc Dissertation, Universidade Federal de Santa Catarina, Florianópolis, Brazil.
- Shimizu, J.Y. and A.R. Higa. 1980. Variação genética entre e dentro de populações de *Araucaria angustifolia* (Bert.) O. Ktze na região de Itapeva – São Paulo. Meeting on Forestry Problems of the Genus *Araucaria*. Fundação de Pesquisas Florestais do Paraná (FUPEP), Curitiba, Brazil.
- Shimizu, J.Y. and Y.M.M. Oliveira. 1981. Distribuição da variação e usos dos recursos genéticos de araucária no Sul do Brasil. EMBRAPA-URPFCS (Empresa Brasileira de Pesquisa Agropecuária). Curitiba, Brazil.
- Shimizu, J.Y., P. Jaeger and S.A. Sopchaki. 2000. Variabilidade genética em uma população remanescente de araucária no Parque Nacional do Iguaçu, Brasil. *Boletim de Pesquisa Florestal* 41:18-36.
- Shimoya, S. 1962. Contribuição ao estudo do ciclo biológico de *Araucaria angustifolia* (Bertoloni) O. Ktze. *Experientiae* 2(2):520-540.
- Turra, F.E. 2003. Cooperativismo Paranaense. Curitiba, Brazil.

Chapter 8

Genetic and ecological aspects of nonwood forest product exploitation in two western Amazonian settlements

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1. Introduction

This chapter provides an overview of the evolution of land use and land tenure regimes in the Brazilian Amazon, illustrated by examples from Acre State. It then assesses the genetic and ecological characteristics of four nonwood forest product (NWFP) species in two types of rural settlements: Settlement Projects (Projetos de Assentamento or PAs) and Extractive Settlement Projects (Projetos de Assentamento Extrativista or PAEs), as defined by INCRA (Instituto Nacional de Colonização e Reforma Agrária or National Institute for Colonization and Agrarian Reform). These two types of settlement were chosen because they encompass different tenure and land-use regimes, and therefore might lead to different patterns of impact on natural resources.

We also review the settlement types and why they were created, and then make recommendations for improved forest management practices based on our research findings.

2. Evolution of land use and land tenure regimes in Acre State

In the late 1870s, driven by recurrent drought and encouraged by the government, about 250 000 people from northeastern Brazil (especially from Ceará State) migrated to the Amazonian region to work in the emerging rubber industry. The Amazonian State of Acre (ca. 150 000 km²) in particular was subjected to significant rates of immigration. Following the collapse of rubber activity in the early 1900s, immigration dropped. However because of an increase in demand for rubber during World War II, the flow of migrants from northeastern Brazil began again.

Starting around 1970, the federal government of Brazil launched a series of regional development programmes and agrarian reform initiatives. Among these, the National Integration Programme (Programa de Integração Nacional or PIN) in 1970 and the Programme for Land Redistribution and Stimulus of the Agro-industry of the North and Northeast

(Programa de Redistribuição de Terras e de Estímulo à Agroindústria do Norte e Nordeste or PROTERRA) in 1971 received the most attention and resources. With the goal of occupying the area of the Amazon along the Trans-Amazon Highway, PIN created farming settlements (Colonization Projects or PCs) aimed at “integrating the men without land in the northeast with the land without men in Amazônia”.

Beginning in the late 1960s, stimulated by incentives for livestock raising, mining and logging, people and businesses from southern Brazil began moving into Acre. Until the early 1970s, rubber extraction had been the major industry in Acre, where it was practised through the so-called ‘barracões de aviamento’ where owners of forested land exchanged rubber for goods and services. By the 1970s, livestock raising had become widespread in Brazil and was causing tensions between traditional land users (communities, populations and farmers) and the new settlers with different tenure and access regimes.

Between 1970 and 1975, about 80% of the state’s territory was sold to new investors from the south (Brandford and Glock 1985), causing a decrease of about 65% in the area dedicated to ‘extraction’ activities (rubber tapping and Brazil nut gathering). Nevertheless, the number of rural landholdings declaring extraction as their principal economic activity increased, the result of growing numbers of newly autonomous rubber tappers who had remained in the forest after the collapse of the traditional rubber estates (Schwartzman 1992). In the 1980s, under the leadership of Chico Mendes, the rights of rubber tappers were defended in the face of increasingly powerful opposition from the newly created and enlarged farms.

In 1985, the National Council of Rubber Tappers (Conselho Nacional dos Seringueiros – CNS) was created, and it initiated a fight to award concessions in the form of ‘seringais’ (properties designated for rubber extraction) to communities actively involved in forest product extraction in Acre.

At about the same time, the National Plan for Land Reform in Brazil was launched, and it created further types of settlements because of what were deemed to be unsatisfactory results obtained through the earlier PCs (later called *Projetos de Assentamento* – PAs or Settlement Projects). Thus, in 1987, the Federal Agrarian Reform Agency (INCRA) established Extractive Settlement Projects (*Projetos de Assentamento Extrativista* or PAEs) and, following a presidential decree in 1990, the Brazilian Institute for the Environment and Renewable Natural Resources (IBAMA), with the support of the CNS, created Extractive Reserves (*Reservas Extrativistas* – RESEXs; see Box 1). RESEXs emerged as a land use type that was designed to legitimize the forest resource use rights of families traditionally living in forested areas. A new type of Conservation Unit (CU) was thereby created in Brazil in the form of specially designated areas allocated to local people for the sustainable use of natural resources for both internal consumption and trade.

Despite their sometimes conflicting agendas, two federal agencies – one (INCRA) traditionally engaged in opening up forests for human settlement and agricultural development, and the other (IBAMA) trying to protect forest ecosystems and their biological resources – became co-responsible agencies for overseeing extractive activities in Brazilian forests (Smith *et al.* 1995).

In 2000, approximately 17% (or 2 598 886 ha) of Acre State was occupied by PAEs and RESEXs, with about 4500 families estimated to be living on this land. Approximately 72% of these families were living on RESEXs (IBGE 2000). A list of PAEs and RESEXs through 2002 is presented in Table 1 (on p. 152).

Differences between PAs and PAEs

Families involved in the earlier PCs or the later PAs were usually farmers who replaced the forest with crops, thereby securing their tenure rights at the same time. In contrast, the newer PAEs were managed by settlers who had already been in the area for some time, and these projects showed greater sensitivity toward resource conservation. Another significant difference between PAs and PAEs was the size of the plots allocated to the families, and this had implications for deforestation and landscape degradation. PAE plots were usually around 300 ha, while PC and PA plots varied between 25 ha and 80 ha (see Table 1). In the PCs and PAs, crop cultivation usually occurred for periods of one to three years after forest clearance. Annual crops were planted in the first year after slash

Box 1. Extractive Reserves (RESEXs)

RESEXs are protected areas that aim to achieve economic self-sufficiency through the sustainable use of renewable natural resources by people practising traditional extractive systems. The extractive regime is regulated by a concession contract accompanied by a management plan approved by IBAMA (Allegretti 1992).

Extractive reserves are characterized by a complex distribution of individuals and community rights, typically dictated by the spatial distribution patterns of the resources as opposed to the standardized patterns used in other tenure regimes. They have been defined by Allegretti (1990) as “public lands designated for the specific purpose of sustainable use of forest products, with property rights designated according to traditional patterns of land use rather than imported models of occupation”.

The ‘seringal’ is a rubber tree tract or forest that falls in part or wholly in the reserve and is divided into smaller management units (‘colocações’). Each landholding is occupied by one family and contains their rubber trails or estradas de seringa. An average family usually works three trails, each of which could contain as many as 150 rubber trees. The size of the colocação is actually defined by the number of adult *Hevea brasiliensis* trees found within it (CNS 1993). This land allocation system is thus very different from the other land allocation types where plots have less flexibility in their defined geographical extent.

The resource rights in RESEXs are therefore defined initially by the location of the rubber trees and then by the trails that link them. The seringueiros allow others to pass freely through their colocação, but the areas where the trails are located are regarded as relatively exclusive. In addition, each family has a cleared area where they live, cultivate a few subsistence crops and raise small animals (Murrieta and Rueda 1995). Brazil nuts constitute the second most important extractive product, and the trees are generally regarded as being the property of a particular colocação. The seringueiros also exercise hunting rights over designated areas of the forest.

and burn, to be replaced by perennial crops or pastureland the following years. In the PAEs, cultivated areas were much smaller and were often returned to secondary forest once they were abandoned. In the PAs, because the forest clearings were relatively larger than those in the PAEs, and because they were used for livestock grazing, they were also more likely to become seriously degraded. Because of these intrinsic differences, the impact of the PC and PA schemes on forest stands is substantially greater than that of PAEs.

In Acre State, natural resource use conflicts occurred in areas where PAs bordered PAEs, and when PA farmers exerted steadily increasing pressure on the forest resources surrounding their settlements through hunting and collection of NWFPs. Despite these conflicts, Acre has the lowest deforestation rate among the Brazilian Amazon states (Lorentzen and Amaral 2002) and this has been interpreted as evidence that frameworks regulating access to and use of natural resources create good conditions for the conservation of forest ecosystems.

A new natural resources management policy designed to improve local standards of living was introduced in 1999 in Acre State. Under this policy, the state government (called the Governo da Floresta or the Government of the Forest) encourages certified extraction of timber and NWFPs, and at the same time supports traditional agriculture. Under the new directives, forest exploitation is regulated by specific agreements and contracts between the state and third parties, whether these be private firms, nongovernmental organizations (NGOs), traditional population associations and their representative institutions like the National Centre for the Sustainable Development of Traditional Populations (Centro Nacional de Desenvolvimento Sustentado das Populações Tradicionais – CNPT) or others.

Table 1. Extractive Reserves (RESEXs) and Extractive Settlement Projects (PAEs) in Amazônia, presented in chronological order of establishment.

Projects	Location	Creation	Area (ha)	Population
INCRA PAEs				
Remanso	Capixaba/AC	1987	39 570	790
Santa Quitéria	Brasiléia	1988	44 205	750
Cachoeira -Chico Mendes	Xapuri/AC	1988	24 898	340
Antimary	Boca do Acre/AM	1988	260 227	4 340
Terruã	Pauini/AM	1989	139 236	2 320
Riozinho	Sena Madureira/AC	1989	35 896	600
Porto Dias	Acrelândia/AC	1989	22 125	415
Porto Rico	Epitaciolândia/AC	1991	7 530	230
Santo Antonio Mourao	Eirunepé/AM	1992	21 525	1 000
Maracá I, II, III	Mazagão/AP	1997	363 500	5 340
Praialta/Piranheira	Nova Efigênia/PA	1997	22 000	1 250
Canary	Bujari/AC	1997	8 054	135
Caquetá (*)	Porto Acre/AC	1997	28 686	516
Limoeiro	Bujari/AC	1998	11 150	185
Subtotal			1 028 602	18 211
IBAMA RESEXs				
Alto Juruá	Thaumaturgo/AC	1990	506 186	~6 000
Chico Mendes	Various counties/AC	1990	970 570	~9 000
Rio Cajari	Various counties/AP	1990	481 650	3 283
Rio Ouro Preto	Guajará-Mirim/RO	1990	204 583	431
Quilombo do Freixal	Mirinzal/MA	1992	9 542	900
Ciriáco	Cindelândia/MA	1992	7 050	1,150
Mata Grande	Senador LaRoque/MA	1992	10 450	500
Médio Juruá	Carauari/AM	1997	253 226	700
Tapajós-Arapiuns	Santarém/PA	1998	647 610	4 000
Lago do Cuniã	Porto Velho/RO	1999	52 065	400
Tarauacá	Tarauacá/AC	2000	151 199	n.a.
Cazumbá-Iracema	Sena Madureira/AC	2002	748 817	1 300
Subtotal			4 052 228	~27 000
TOTAL			5 080 830	~ 45 211

Source: adapted from INCRA and IBAMA, <http://www.ibama.gov.br/resex/resex.htm> (Brown and Resende 2000; Kainer *et al.* 2003). Explanation: AC = Acre, PA = Pará, RO = Roraima, MA = Maranhao, AP = Macapá, AM = Amazonas. (*) Seringal Caquetá is listed among PAEs, although at the time the study was carried out it was not yet officially recognized as such, although it was managed as a PAE.

3. Research activities

3.1 The study sites

We selected study sites in the Brazilian State of Acre (Figure 1), in the municipality of Porto Acre. Officially established in 1992, this municipality covers 2923 km² and has a population of 9749 people, 88% of whom live in rural areas. Within these sites, we investigated one PA (Porto Alonso) and one PAE (Caquetá).

We selected four tree species for our study based on their importance to local livelihoods, whether this was for their commercial value or for local consumption: *Bertholletia excelsa*, *Hevea brasiliensis*, *Carapa guianensis* and *Euterpe precatoria*. These species have different ecological traits and spatial distributions and therefore permitted good assessments of the impact of differential forest management. The principal characteristics of these species are presented in Table 2 (overleaf).

Although the forest stands studied were subjected to NWFP extraction, no signs of clear cutting were visible. In the PA scheme, forested areas may be partially disturbed by their proximity to areas where the forest has been cleared or fragmented for conversion into agriculture. Despite the absence of recent deforestation, evidence of removal of individuals, especially of *C. guianensis* and *E. precatoria*, was found in the transects established in the Porto Alonso PA. Owing to the very small population size of *B. excelsa* found in the Caquetá PAE, sampling of this species was extended to another area called Colocação Rio de Janeiro, a part of the Chico Mendes extractive reserve (a RESEX), the largest in Brazil (about 970 000 ha), located in the southeastern corner of the state of Acre and stretching across five municipalities.

3.2 Methods

With a special focus on NWFP extraction patterns and on income levels derived from NWFPs, household surveys were conducted in the two communities of Porto Alonso PA and Caquetá PAE in order to understand the past and current patterns of forest resource exploitation, and thereby to assist in the interpretation of results from the ecological and genetic analyses of the four selected species. The survey also enabled us to identify the main stakeholders and groups of forest users.



Figure 1. Location of Acre State and its main towns.

Table 2. Characteristics of *B. excelsa*, *H. brasiliensis*, *E. precatoria* and *C. guianensis*.

Description	<i>Bertholletia excelsa</i>	<i>Hevea brasiliensis</i>	<i>Euterpe precatoria</i>	<i>Carapa guianensis</i>
Portuguese names	Castanheira	Seringueira	Açaí	Andiroba
Height (m), dbh* (cm)	30–40 m, 100–400 cm	20–30 m, 30–80 cm	10–20 m, 6–40 cm	15–30 m, 30–120 cm
Successional group	Late secondary	Early to late secondary	Climax	Late secondary to climax
Distribution	Endemic to the Amazon	Endemic to the Amazon	Very common (100 adults/ha), mostly in floodplain	Common (7 adults/ha in dry areas; 20 adults/ha in the floodplain)
Leaf-change	Semi-evergreen to deciduous	Semi-evergreen	Evergreen	Evergreen
Sex	Monoecious	Monoecious	Monoecious	Monoecious
Flowers	Bisexual	Unisexual, on the same inflorescence	Unisexual	Bisexual
Flowering	November – February	August – November	September – December	August – October
Pollination	Insects (bees)	Insects (three species of fly in Heleidae)	Coleoptera and small insects, cross pollinated	Bees or beetles
Fruit maturation	December – March	April – May	April – August	January – April
Seed	Long and hard, 4–7 cm	Big and round, 3–5 cm	Round, 0.9–1.3 cm	Four-cornered nut, 3–5 cm
Seed dispersion	Wind, barochory (unassisted)	Hydrochory (by water)	Birds, rodents	Hydrochory, barochory, rodents
Seed bank	No	No	No	No

* dbh = diameter at breast height.

Two series of plots were established to investigate the diameter and spatial distribution of the four selected species: (i) five transects of 1 ha each were established in Colocação Limoeiro I, inside Caquetá (PAE); (ii) three transects of 0.8 ha each were created in Colônia Santa Luzia, inside Porto Alonso (PA). Adult trees (>30 cm dbh) and younger trees were sampled (only adults were sampled for *E. precatoria*). Seedlings and saplings were grouped into three size categories: juvenile I (individuals between 0.5–1.0 m in height); juvenile II (between 1.0–2.0 m in height); and juvenile III (above 2.0 m in height though still immature). Spatial distribution maps were produced for the adult trees of *B. excelsa*, *C. guianensis* and *E. precatoria*; *H. brasiliensis* was mapped in Caquetá (PAE) along a trail opened by rubber tappers. The number of individuals sampled for the four species is presented in Table 3.

Owing to the variable size of samples, different genetic parameters were studied for each species, as follows:

Genetic features	<i>B. excelsa</i>	<i>H. brasiliensis</i>	<i>E. precatoria</i>	<i>C. guianensis</i>
Genetic diversity	X	X	X	X
Genetic structure	X	X	X	X
Outcrossing rate	-	X	-	-
Gene flow	-	X	X	X

Microsatellite and random amplified polymorphic DNA (RAPD) techniques were used for the genetic analyses, and were selected based on the availability of protocols and markers from previous studies. Microsatellites were used for *E. precatoria*, *C. guianensis* and *H. brasiliensis* (see Table 4, overleaf), and RAPDs were used for *B. excelsa* because simple sequence repeats or microsatellites (SSRs) were not available. DNA extraction was performed according to the cetyltrimethylammonium bromide (CTAB) protocol (Ferreira and Grattapaglia 1995).

Table 3. Number of adult and juvenile individuals sampled for four tree species. The *B. excelsa* individuals sampled from Colocação Rio de Janeiro, RESEX Chico Mendes (municipality of Xapuri) were not included in the calculation of means because they were located outside the transect.

Species	Population	Type of settlement	Adults	Juveniles			Total
				I	II	III	
<i>H. brasiliensis</i>	Limoeiro I	PAE	32	26	22	23	103
	Santa Luzia	PA	8	23	6	14	51
<i>E. precatoria</i>	Limoeiro I	PAE	27	25	25	25	102
	Santa Luzia	PA	10	25	14	19	68
<i>C. guianensis</i>	Limoeiro I	PAE	33	24	18	25	100
	Santa Luzia	PA	14	10	3	6	33
<i>B. excelsa</i>	Limoeiro I	PAE	34	-	2	15	51
	Rio de Janeiro	RESEX	23	-	-	-	23
Average per population			20.67	22.17	14.67	18.67	
Total							531

Table 4. Primer SSRs used for each species, number of alleles, allelic range* and source of information.

Species	SSR	No. of alleles	Allelic range	Source
<i>C. guianensis</i>	CG 05	15	164–200	Dayanandan <i>et al.</i> 1999
	CG 07	13	202–228	
<i>E. precatoria</i>	EE 08	12	96–122	Gaiotto <i>et al.</i> 2001; originally for <i>E. edulis</i>
	EE 45	14	104–140	
	EE 54	7	116–138	
	EE 59	5	96–122	
<i>H. brasiliensis</i>	AGHE21	13	270–302	Lespinasse <i>et al.</i> 2000
	AGHE51	11	248–268	
	AGHE56	13	184–234	
	AGHE67	11	102–122	
	AGHE69	21	220–262	
	AGHE175	8	164–180	
	AGHE185	5	94–112	

* Microsatellite loci are characterized by the repetition of a fundamental motif comprising a short sequence of nucleotides. Alleles at a microsatellite locus differ in the number of repetitions of this fundamental motif called the repeat number or allele size. The minimum and maximum number of repetitions provides the allelic range. This value is a key parameter that should be properly estimated in order to proceed with calculations of divergence times in phylogenetic studies and to better investigate the within- and between-population variability.

Genetic diversity was assessed in juveniles and adults using the following parameters:

- allelic richness or mean number of alleles by loci (A),
- effective number of alleles by loci ($n_e = 1/\sum p_i^2$ where p_i is the frequency of the i^{th} allele),
- expected heterozygosity (H_e), and
- observed heterozygosity (H_o).

The lack of heterozygote deficiency was evaluated by calculating the fixation index ($f = 1 - H_o/H_e$). The genetic diversity between sites and size classes was estimated by bootstrapping across loci, using the software programs GDA (Lewis and Zaykin 2000) and GenAlEx (Peakall and Smouse 2001, <http://www.anu.edu.au/BoZo/GenAlEx>).

Genetic structure was determined using the F -statistic and calculated using the formula developed by Weir and Cockerham (1984) that identifies a genetic structure by partitioning variation through an analysis of variance. The parameters estimated were:

- mean coefficient of inbreeding within the populations (f),
- mean total coefficient of inbreeding of the species, or Wright's coefficient (F) and
- mean co-ancestry coefficient indicating genetic drift among subpopulations (θ_p).

Confidence intervals at 99% probability were estimated by bootstrapping across loci, using 10 000 replications. The spatial structure was analyzed through the multiallelic spatial autocorrelation analysis, applied to adult populations in Colocação Limoeiro I (Smouse and Peakall 1999) using the GenAlEx program.

The Multilocus Mating System Program (MLTR) developed by Ritland (1996) was used for *H. brasiliensis* to estimate outcrossing rate based on: multiple loci (t_m); the outcrossing rate for each individual locus by obtaining mean estimates for single populations (t_s); Wright's coefficient of inbreeding (F); the correlation between seed pairs (r_s); and the correlation between pollen grains of the outcrossed seed pairs (r_p).

Gene flow (N_m) was analyzed in *C. guianensis*, *E. precatoria* and *H. brasiliensis*, and was calculated indirectly through the estimated values of F_{ST} (genetic diversity among populations) according to the following model:

$$N_m = (1 - F_{ST}) / 4 F_{ST}$$

3.3 Results – analysis of forest stand structure

Between the two study sites, the distribution of size classes was similar for *B. excelsa* and *C. guianensis*, but there was a statistically significant difference in the number of individuals of *E. precatoria* and *H. brasiliensis* in each size class (Figure 2). Only *H. brasiliensis* showed a typical inverted 'J' distribution of diameters, where the number of individuals progressively decreases with increasing size – normally expected when continuous regeneration of a species occurs. While there were a limited number of medium- and large-sized juveniles of *C. guianensis*, *B. excelsa* occurred at too low a density to allow us to detect any size structure within the transects investigated. Despite very high density of individuals, *E. precatoria* lacked small- and intermediate-sized juveniles, indicating constraints in the regeneration of the species.

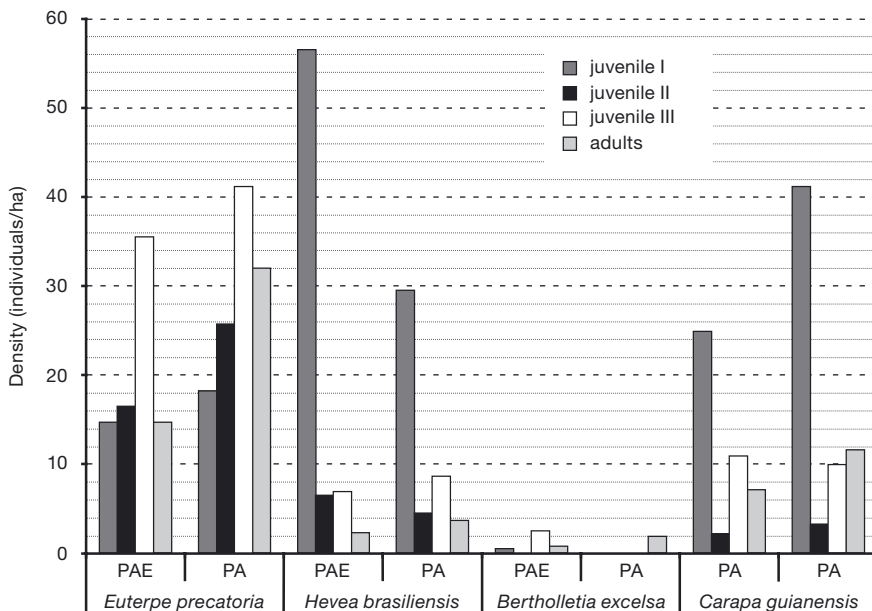


Figure 2. Density (trees/ha) of *E. precatoria*, *H. brasiliensis*, *B. excelsa* and *C. guianensis* in the following size classes: juvenile I, juvenile II, juvenile III and adult, in the transects established in Colocação Limoeiro I (Caquetá PAE) and in the Colônia Santa Luzia (Porto Alonso PA).

3.4 Results – analysis of the forest genetic study

Carapa guianensis

Two populations were analyzed in Colocação Limoeiro I (Caquetá PAE) and in Colônia Santa Luzia (Porto Alonso PA), and compared. Owing to the much smaller size of the sample from Porto Alonso PA, the juveniles sampled at this site were lumped together in one size class. Microsatellite results from the two loci are presented in Table 5. At Colocação Limoeiro I, 24 alleles were identified (12 alleles per locus). The results showed that the number of alleles per locus, or allelic richness, did not differ significantly between size classes. Allelic richness (A) estimates were found to be consistent with results from other studies of the same species in Costa Rica (Dayanandan *et al.* 1999). Estimates of the effective number of alleles per locus (n_e) were always lower than the values of allelic richness (A). In our view, this indicates an uneven distribution of allele frequencies, indicating that many low-frequency alleles are more likely to be lost if the processes of bottleneck and genetic drift occur.

The values of expected heterozygosity (H_e) were high and did not differ significantly among size classes, showing that high gene diversity is maintained throughout the generations.

We found that the Colônia Santa Luzia (Porto Alonso PA) population of *C. guianensis*, which was more heavily exploited, still maintains values of allelic richness similar to the less-exploited population at Caquetá PAE, although the population studied from the Porto Alonso PA shows more signs of inbreeding (Table 6). This inbreeding may be due to a smaller number of mature trees contributing to mating or to self-fertilization. But assumptions about the effects of reproductive biology on the genetic parameters of *C. guianensis* are difficult to prove because so little is known about the mating system of the species. Although the high level of inbreeding in *C. guianensis* at Porto Alonso PA is apparently a recent occurrence, it nonetheless poses a risk to the maintenance of long-term genetic diversity.

Table 5. Genetic parameters of a *C. guianensis* population from Colocação Limoeiro I (Caquetá PAE) and from Colônia Santa Luzia (Porto Alonso PA). Two SSR loci were analyzed in juveniles and adults.

Limoeiro I (Caquetá PAE)	n	A	n_e	H_e	H_o	f
Adults	29.5 (0.71)	7.50 (2.12)	4.37 (1.05)	0.76 (0.06)	0.65 (0.21)	0.16 (0.21)
Juveniles I	21.0 (0.00)	7.00 (4.24)	3.98 (0.80)	0.74 (0.05)	0.55 (0.50)	0.28 (0.63)
Juveniles II	16.5 (2.12)	9.50 (2.12)	4.70 (2.00)	0.76 (0.10)	0.67 (0.38)	0.15 (0.39)
Juveniles III	23.5 (0.71)	8.00 (0.00)	4.76 (0.63)	0.79 (0.03)	0.49 (0.13)	0.38 (0.15)
Santa Luzia (Porto Alonso PA)	n	A	n_e	H_e	H_o	f
Adults	11.0 (1.44)	7.00 (1.40)	4.15 (1.02)	0.75 (0.06)	0.52 (0.26)	0.32 (0.29)
Juveniles	18.0 (0.00)	8.50 (2.12)	3.77 (2.37)	0.69 (0.21)	0.53 (0.11)	0.20 (0.07)

n = number of individuals analyzed, A = allelic richness, or number of alleles per locus, n_e = effective number of alleles per locus, H_e = expected heterozygosity, H_o = observed heterozygosity and f = fixation index. Values in parentheses are standard deviations.

Table 6. Parameters of genetic structure for four populations (two adult and two juvenile) of *C. guianensis*, from both Colocação Limoeiro I (Caquetá PAE) and Colônia Santa Luzia (Porto Alonso PA).

Parameter	Adults		Juveniles	
	Estimated value	Confidence interval	Estimated value	Confidence interval
f	0.2054	0.0494–0.3920	0.2800	0.0920–0.5029
F	0.2178	0.0509–0.4114	0.2988	0.0896–0.5318
θ_p	0.0156	0.0016–0.0319	0.0262	-0.0026–0.0582
N_m	15.77		9.29	

A confidence interval of 99% was obtained through 10 000 bootstraps over loci. f : mean coefficient of inbreeding within populations; F : mean total coefficient of inbreeding of a species; θ_p : coefficient of genetic drift; N_m : gene flow.

The mean values of inbreeding caused by the mating system (f) were similar to those for total inbreeding (F). This seems to indicate that most of the inbreeding is caused by the mating system and not by genetic divergence among populations. The estimates of θ_p were very low, showing that most of the genetic diversity is found within populations, and gene flow is counterbalancing the effects of genetic drift.

In the juveniles, the values of these parameters were similar to those found in adults (Table 6).

Hevea brasiliensis

Table 7 presents the outcomes of the genetic analyses carried out on *H. brasiliensis* at the two sites. Levels of H_e found at both sites were very high and comparable to data obtained on the species in Acre using other markers ($H_e = 0.45$ using RFLP – restriction fragment length polymorphism – markers, Besse *et al.* [1994]; and $H_e = 0.60$ using isozymes, Chevallier [1988] cited in Besse *et al.* [1994]).

Table 7. Genetic parameters of two *H. brasiliensis* populations from Limoeiro I (Caquetá PAE) and Santa Luzia (Porto Alonso PA). Two SSR loci were analyzed in juveniles and adults.

Limoeiro I (Caquetá PAE)	n	A	n_e	H_e	H_o	f
Adults	30.71 (1.25)	7.86 (4.91)	4.26 (3.85)	0.62 (0.25)	0.55 (0.26)	0.11
Juveniles I	25.71 (0.49)	6.43 (3.69)	3.02 (1.70)	0.58 (0.22)	0.44 (0.30)	0.24
Juveniles II	20.86 (1.07)	6.71 (2.63)	3.78 (2.69)	0.61 (0.23)	0.61 (0.28)	0.00
Juveniles III	21.71 (1.25)	6.28 (3.50)	3.07 (1.51)	0.61 (0.16)	0.50 (0.28)	0.18
Santa Luzia (Porto Alonso PA)	n	A	n_e	H_e	H_o	f
Adults	7.71 (0.49)	4.57 (2.37)	3.57 (2.25)	0.61 (0.24)	0.53 (0.30)	0.13
Juveniles I	22.28 (1.11)	6.57 (2.07)	3.01 (1.66)	0.59 (0.18)	0.58 (0.29)	0.01
Juveniles II/III	18.86 (1.57)	6.00 (3.56)	3.60 (2.54)	0.59 (0.29)	0.56 (0.30)	0.05

n = number of individuals analyzed, A = number of alleles per locus, n_e = effective number of alleles per locus, H_e = expected heterozygosity, H_o = observed heterozygosity, and f = fixation index. Values in parentheses are standard deviations.

Low levels of inbreeding (f) were detected in *H. brasiliensis* at both sites. Surprisingly, the estimated fixation index for the juveniles in the more disturbed site of Colônia Santa Luzia is smaller than at Colocação Limoeiro I. The genetic divergence among adult populations due to forest fragmentation ($\theta_p = 0.026$) was not significant (Table 8), suggesting that gene flow among populations is buffering genetic drift.

A t_m estimate of 0.98 indicates that outcrossing is preferred in this species (Table 9). A difference between t_m and t_s of 0.13 suggests that biparental inbreeding is common. The r_p value was 0.531 and r_t value was 0.390, and even though the outcrossing rate was high, these results indicate a high probability of descendants from the same father and mother in a family (r_p). These values were higher than expected for populations with free pollination, indicating that the occurrence of descendants from the same father in maternal families is relatively common. Autofertilization was also observed for some families.

Euterpe precatoria

One hundred and seventy *E. precatoria* trees were analyzed from both sites (Table 10). Genetic diversity (H_e) varied from 0.41 to 0.58. Allelic richness did not differ significantly among size classes, either within the same population or between populations. Both the estimated genetic diversity (H_e) and the observed heterozygosity (H_o) were lower than those found for other species in the genus *Euterpe* (Gaiotto 2001; Souza 2002).

The fixation indices seem to indicate that the screened populations have genotypic characteristics close to that expected in Hardy-Weinberg equilibrium conditions, where mating is random, and there is neither migration nor mutation.

Table 8. Parameters of genetic structure for four populations (two adult and two juvenile) of *H. brasiliensis*, from both Colocação Limoeiro I (Caquetá PAE) and Colônia Santa Luzia (Porto Alonso PA). A confidence interval of 99% was obtained through 10 000 bootstraps over loci.

Parameter	Adults		Juveniles	
	Estimated value	Confidence interval	Estimated value	Confidence interval
f	0.1435	-0.0446–0.2906	0.1393	-0.048 –0.3634
F	0.1655	-0.0411–0.3343	0.1612	-0.0170–0.3732
θ_p	0.0256	-0.0149–0.0958	0.0255	0.0110–0.0372
N_m	9.52		9.55	

Table 9. Estimates of mating system parameters in *H. brasiliensis*.

Parameter	Estimated value*
t_m (multilocus outcrossing rate)	0.980 (0.014)
t_s (outcrossing rate based on the average of individual loci)	0.850 (0.044)
$t_m - t_s$	0.130 (0.046)
r_p (correlation of paternity within progenies originated from outcrossing)	0.531 (0.086)
r_t (correlation between outcrossing rates within families)	0.390 (0.043)

*Values in parentheses are standard deviations estimated through 10 000 bootstraps.

Table 10. Genetic characteristics of two *E. precatoria* populations from Limoeiro I (Caquetá PAE) and Santa Luzia (Porto Alonso PA). Two SSR loci were analyzed in juveniles and adults.

Limoeiro I (Caquetá PAE)	n	A	n_e	H_e	H_o	f
Adults	26.0 (2.0)	4.75 (3.59)	2.85 (2.58)	0.45 (0.34)	0.50 (0.36)	0.11
Juveniles I	24.5 (1.0)	4.75 (1.50)	2.24 (0.50)	0.53 (0.10)	0.54 (0.18)	-0.01
Juveniles II	24.5 (1.0)	4.75 (1.26)	2.62 (0.92)	0.58 (0.14)	0.62 (0.15)	-0.07
Juveniles III	21.5 (5.2)	4.50 (1.29)	2.55 (1.31)	0.54 (0.17)	0.63 (0.26)	-0.17
Santa Luzia (Porto Alonso PA)	n	A	n_e	H_e	H_o	f
Adults	9.75 (0.50)	3.50 (0.58)	1.93 (1.24)	0.41 (0.21)	0.43 (0.31)	-0.04
Juveniles I	23.75 (0.50)	4.75 (1.50)	2.23 (1.60)	0.41 (0.26)	0.40 (0.28)	0.02
Juveniles II	13.50 (1.00)	5.25 (2.50)	2.58 (2.36)	0.42 (0.28)	0.42 (0.39)	0.00
Juveniles III	17.50 (2.38)	4.75 (2.21)	2.13 (0.98)	0.47 (0.19)	0.44 (0.23)	0.06

n = number of individuals analyzed, A = number of alleles per locus, n_e = effective number of alleles per locus, H_e = expected heterozygosity, H_o = observed heterozygosity and f = fixation index. Values between parentheses are standard deviations.

Values of total inbreeding (F) and of inbreeding related to the mating system (f) were near zero (Table 11). Total inbreeding (F) in adults was lower than that found in younger generations. Although juvenile populations still maintain high values of allelic richness, gene flow among populations is not enough to buffer genetic drift.

Bertholletia excelsa

RAPDs were used to analyze the spatial distribution of genetic diversity in *B. excelsa* populations subject to different use regimes from Colocação Limoeiro I (Caquetá PAE) and from Colocação Rio de Janeiro (RESEX Chico Mendes, Xapuri; Table 3) in order to

Table 11. Parameters of genetic structure for four populations (two adult and two juvenile) of *E. precatoria*, from both Colocação Limoeiro I (Caquetá PAE) and Colônia Santa Luzia (Porto Alonso PA). A confidence interval of 99% was obtained through 10 000 bootstraps over loci.

Parameter	Adults		Juveniles	
	Estimated value	Confidence interval	Estimated value	Confidence interval
f	-0.0789	-0.3582–0.0860	-0.0079	-0.1503–0.1129
F	-0.0096	-0.1967–0.1414	0.1432	0.0029–0.1975
θ_p	0.0643	0.0006–0.1189	0.1499	0.0663–0.2035
N_m	3.64	-0.3582–0.0860	1.42	

compare findings with those obtained from other species. Seven primers were used to amplify 39 markers. Among these, 82% were polymorphic in trees from both locations, indicating high genetic variability for the species (Table 12). Higher polymorphism was found in the less-disturbed population at Rio de Janeiro, but the difference was not significant.

The polymorphism recorded in our study was greater than that found in other genetic studies of this species from Acre. For instance, Pardo (2001) and Buckley *et al.* (1988) found 40% and 54.3% polymorphic allozyme loci respectively. However, this could be partly explained by the type of markers used, as RAPD markers tend to be biased towards lower values of expected heterozygosity and greater differentiation between populations (Isabel *et al.* 1995). Thus, this comparison with data from earlier studies has only an indicative value.

4. Concluding remarks

There were no signs that the species investigated here had suffered adverse genetic events, despite undergoing reductions in population size that would normally be expected to cause substantial loss of genetic diversity. This is because heavy forest fragmentation had not yet occurred in the areas investigated in this study. Nevertheless, recent removal of individual trees was found to have left genetic traces in the study area located in the Porto Alonso PA.

Levels of inbreeding were detected in *C. guianensis* in Porto Alonso PA and in *H. brasiliensis* at both sites. Moreover, inbreeding was found in adult and young individuals of both species, indicating that both are sensitive to forest disturbance from fragmentation and extraction of NWFPs.

High inbreeding levels can be linked to constraints in gene flow. Gene flow vectors are essential ecological factors that can determine population genetic structure in species. For example, in tropical tree species pollen flow contributes more to total gene flow than does seed dispersal when seed dispersal is limited (Hamrick *et al.* 1993; Boshier *et al.* 1995), resulting in a different shape to the species' spatial genetic structure. Bawa (1974) has shown that tropical tree species are pollinated mostly by animals. Because deforestation and other forest disturbances can cause an alteration in the behaviour of pollinating agents, induced by lowered density of flowering trees, reduced pollen movement among populations can occur (Aizen and Feinsinger 1994). Forest disturbance may also decrease the abundance of pollinating agents, leading to a reduction in the number of fruits and seeds (Aizen and Feinsinger 1994) and lower outcrossing rates (Kearns *et al.* 1998).

The ecological and reproductive characteristics of the four species studied were different, and these differences partly explain the variable genetic responses to forest disturbance recorded. Because of their differential gene flow characteristics, species that occur in very low densities (e.g., <1 individual/ha) respond differently to fragmentation

Table 12. Polymorphism of RAPD markers from *B. excelsa* trees located at Colocação Limoeiro I and Colocação Rio de Janeiro.

Population		n	Polymorphism (%)
Colocação Limoeiro I (Caquetá PAE)	Juveniles	14	71.79
	Adults	31	71.79
Colocação Rio de Janeiro (RESEX Chico Mendes)	Adults	22	82.05
Total		67	82.00

n = number of individuals sampled.

than do common species. *E. precatoria* and *C. guianensis* occur at high (100 adult trees/ha) and moderate densities (20 adult trees/ha) (Vasconcellos Gama *et al.* 2003), while *H. brasiliensis* and *B. excelsa* are species with low population densities and may be genetically more susceptible to fragmentation.

Genetic studies have shown that rare tropical tree species usually have less genetic diversity than common tropical tree species (Billington 1991). Our study showed that the species with the highest density, *E. precatoria*, also exhibited more pronounced genetic divergence between juveniles and adults. Nevertheless, the expected difference in genetic diversity between rare and common tree species was not observed in our study. While the four species still presented high allelic richness and genetic diversity in adults as well as in juveniles, levels of inbreeding detected in this study and the genetic divergence noted between populations might eventually lead to lower heterozygosity levels and, therefore, to future loss of alleles.

In the short term, loss of heterozygosity can reduce individual fitness, diminishing the viability of remnant populations. In the long term, reduced allelic richness can limit the ability of species to respond to changes in selective pressure (Saunders *et al.* 1991; Frankel *et al.* 1995; Young *et al.* 1996) and ultimately can lead to extinction.

The long-term survival of fragmented populations of tree species depends on the existence of adequate strategies for the management of forest remnants, supported by the analysis of spatial distributions of diversity and the monitoring of changes in genetic processes. A deeper understanding of the biological and genetic processes of a few model species like those presented here is fundamental to our ability to understand the evolutionary potential of forest trees and the conservation needs of forest ecosystems.

Creating a market for important tree species and their products can generate income and also stimulate their conservation. A socioeconomic survey of communities living at our study sites investigated the possibility of trading forest products that traditionally have not been commercialized. We found that the exploitation of NWFPs caused less destruction to forest resources than logging or the creation of pastureland, but timber harvesting and cattle raising still seem to be more remunerative. Thus there is still a risk that people will shift from less-damaging extractive activities to forest clearance.

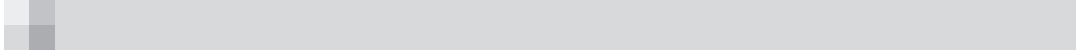
The creation of markets for new NWFPs from *C. guianensis* and *E. precatoria*, to complement those that already exist for *H. brasiliensis* and *B. excelsa*, could be a solution to the continuing deforestation. However, these actions require investment to maintain the forest cover promoted by the rubber and Brazil nut extractive systems, and which is in contrast to the heavy clearing being practised in many of the rural settlements. The exploitation of species like *C. guianensis* and *E. precatoria*, that provide non-traditional NWFPs, should be supported by genetic studies that enable us to define thresholds for their long-term sustainable exploitation.

References

- Aizen, M.A. and P. Feinsinger. 1994. Forest fragmentation, pollination and plant reproduction in a Chaco dry forest, Argentina. *Ecology* 75:330-351.
- Allegretti, M.H. 1990. Extractive reserves: an alternative for reconciling development and environmental conservation in Amazonia. Pp. 252-264 in *Alternatives to Deforestation: Steps Toward Sustainable Use of the Amazon Forest* (A. Anderson, ed.). Columbia University Press, New York, USA.
- Allegretti, M.H. 1992. Reconciling people and land: The prospects for sustainable extraction in the Amazon. Pp. 249-254 in *Development or Destruction? The Conversion of Tropical Forest to Pasture in Latin America* (T.E. Downing, S. Hecht, H.A. Pierson and G. Garcia-Downing, eds.). Westview Press, Boulder, USA.
- Bawa, K.S. 1974. Breeding systems of tree species of a lowland tropical community. *Evolution* 28:85-92.
- Besse, P., M. Seguin, P. Lebrun, M.H. Chevallier, D. Nicolas and C. Lanaud. 1994. Genetic diversity among wild and cultivated populations of *Hevea brasiliensis* assessed by nuclear RFLP analysis. *Theor. Appl. Gen.* 88:199-207.

- Billington, H.L. 1991. Effect of population size on genetic variation in a dioecious conifer. *Conserv. Biol.* 5:115-199.
- Boshier, D.H., M.R. Chase and K.S. Bawa. 1995. Population genetics of *Cordia alliodora* (Boraginaceae), a neotropical tree. Gene flow, neighbourhood, and population substructure. *Am. J. Bot.* 82:484-490.
- Brandford, S. and O. Glock. 1985. *The Last Frontier: Fighting Over Land in the Amazon*. Zed Books, London, UK.
- Brown, K. and S. Resende. 2000. Environmentalists, rubber tappers and empowerment: the politics and economics of extractive reserves. *Dev. Change* 31(1):201-27.
- Buckley, D.P., D.M. O'Malley, V. Apsit, G.T. Prance and K.S. Bawa. 1988. Genetics of Brazil nut (*Bertholletia excelsa* Humb. and Bonpl.: Lecythidaceae). 1. Genetic variation in natural populations. *Theor. Appl. Gen.* 76:923-928.
- Chevallier, M.H. 1988. Genetic variability of *Hevea brasiliensis* germplasm using isozyme markers. *J. Nat. Rubb. Res.* 3(1):42-53.
- CNS (Conselho Nacional dos Seringueiros). 1993. *Poronga Comunicação e Editoração Ltda. Diretrizes para um programa de reservas extrativistas na Amazônia*. Rio Branco, Brazil.
- Dayanandan, S., J. Dole, K.S. Bawa and R. Kesseli. 1999. Population structure delineated with microsatellite markers in fragmented populations of a tropical tree, *Carapa guianensis* (Meliaceae). *Mol. Ecol.* 8:1585-1593.
- Ferreira, M.E. and D. Grattapaglia. 1995. *Introdução ao uso de Marcadores Genéticos* (2nd edn.) EMBRAPA-CENARGEN (Empresa Brasileira de Pesquisa Agropecuária-Centro Nacional de Recursos Genéticos e Biotecnologia), Brazil.
- Frankel, O.H., A.H.D. Brown and J.J. Burdon. 1995. *The Conservation of Plant Biodiversity*. Cambridge University Press, Cambridge, UK.
- Gaiotto, F.A. 2001. *Inferências sobre herança quantitativa e estrutura genética em populações naturais de Euterpe edulis Mart. Utilizando marcadores microsatélites*. Piracicaba. PhD thesis, Escola Superior de Agricultura Luis de Queiroz, University of São Paulo, Brazil.
- Gaiotto, F.A., R.P.V. Brondani and D. Grattapaglia. 2001. Microsatellite markers for Heart of Palm – *Euterpe edulis* and *Euterpe oleracea* Mart. (Arecaceae). *Mol. Ecol.* 1:86-88.
- Hamrick, J.L., D.A. Murawski and J.D. Nason. 1993. The influence of seed dispersal mechanisms on the genetic structure of tropical tree populations. *Vegetatio* 107/108:281-297.
- IBGE (Institute of National Statistics and Geography). 2000. *Censo Demográfico 2000*. IBGE, Rio de Janeiro, Brazil.
- Isabel, N., J. Beaulieu and J. Bousquet. 1995. Complete congruence between gene-diversity estimates derived from genotypic data at enzyme and random amplified polymorphic DNA loci in black spruce. *Proc. Nat. Acad. Sci. USA* 92:6369-6373.
- Kainer, K.A., M. Schmink, A.C.P. Leite and M.J. da Fadell Silva. 2003. Experiments in forest-based development in western Amazonia. *Soc. Nat. Resour.* 16(10):869-886.
- Kearns, C.A., D.W. Inouye and N.M. Waser. 1998. Endangered mutualism: the conservation of plant-pollinator interactions. *Annu. Rev. Ecol. Sys.* 29:83-112.
- Lespinasse, D., M. Rodier-Goud, L. Grivet, A. Leconte, H. Legnate and M. Seguin. 2000. A saturated genetic linkage map of rubber tree (*Hevea* spp.) based on RFLP, AFLP, microsatellite, and isozyme markers. *Theor. Appl. Gen.* 100:127-138.
- Lewis, P. O. and D. Zaykin. 2000. *GDA: genetic data analysis software*. University of Connecticut, Storrs, USA.
- Lorentzen, E. and W.A.N. Amaral. 2002. Desmatamento. Pp. 163169 *in Situação do Meio Ambiente no Brasil: Uma Visão da Sociedade Civil* (A. Camargo, J.P. Capobianco and J.A.P. Oliveira, eds.). FGV/ISA (Fundação Getulio Vargas/Instituto Socioambiental), Brazil.
- Murrieta, J.R. and R.P.R. Rueda. 1995. *Extractive Reserves*, IUCN Forest Conservation Programme, World Conservation Union (IUCN), Gland, Switzerland.
- Pardo, M. 2001. *Estrutura genética de Castanha do Brasil (Bertholletia excelsa H.B.K.) em floresta e em pastagens no leste do Estado do Acre*. Dissertação de Mestrado. Piracicaba, Brasil.

- Ritland, K. 1996. A marker-based method for inferences about quantitative inheritance in natural populations. *Evolution* 50:1062-1073.
- Saunders, D.A., R.J. Hobbs and C.R. Margules. 1991. Biological consequences of ecosystem fragmentation: a review. *Conserv. Biol.* 5(1):18-32.
- Schwartzman, S. 1992. Land distribution and the social costs of frontier development in Brazil: social and historical context of extractive reserves. Pp. 51-66 *in* Non-Timber Products from Tropical Forests. Evaluation of a conservation and development strategy. (D.C. Nepstad and S. Schwartzman, eds.). Advances in Economic Botany. The New York Botanical Garden, New York, USA.
- Smith, N.J.H., E.A.S. Serrão, P.T. Alvim and I.C. Falesi (eds.). 1995. Amazonia: Resiliency and Dynamism of the Land and its People. United Nations University Press, Tokyo, Japan, New York, USA, Paris, France.'
- Smouse, P.E. and R. Peakall. 1999. Spatial autocorrelation analysis of multi-allele and multi-locus genetic micro-structure. *Heredity* 82:561-573.
- de Souza, P.C.A. 2002. Aspectos ecológicos e genéticos de uma população natural de *Euterpe oleracea* Mart. no estuário amazônico. MA thesis, Escola Superior de Agricultura Luis de Queiroz, University of São Paulo, Oiracicaba, Brazil.
- Vasconcellos Gama, J.R., S.A. Botelho, M. de Matos Bentes-Gama and J.R. Soares Scolforo. 2003. Estrutura e potencial futuro de utilização da regeneração natural de floresta de várzea alta no município de Afuá, estado de Pará. *Ciência Florestal* 13(2):71-82.
- Weir, B.S. and C.C. Cockerham. 1984. Estimating *F*-statistics for the analysis of population structure. *Evolution* 38:1358-1370.
- Young, A.G., T. Boyle and T. Brown. 1996. The population genetic consequences of habitat fragmentation for plants. *Trends Ecol. Evol.* 11(10):413-418.



Chapter 9

Conservation of Mata Atlântica forest fragments in the State of São Paulo, Brazil

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1. Introduction

This chapter describes the current status of forest genetic resources (FGRs) in four tree species found in fragments of the Atlantic Forest located in southwestern São Paulo State (Pontal), Brazil, now part of a heavily degraded agricultural landscape. The four species were selected in consultation with local stakeholders, and a genetic assessment was complemented by ecological and socioeconomic surveys on current patterns of FGR use by farmers. The first part of the chapter introduces how historical events led to the currently reduced forest cover in Pontal and to the serious degradation of what was once a highly diverse semideciduous forest biome. The selected species, representative of different guilds with different characteristics, along with details of the genetic and the ecological surveys, are also described. A socioeconomic survey was undertaken to identify the principal forest users and stakeholders and to permit interpretation of the ecological and genetic findings, which are presented in the second part of the chapter.

2. Land use history of the Pontal region

The study area is located in the Pontal do Paranapanema region of western São Paulo State (Figure 1) in what was once the continuous Mata Atlântica forest. Large parts of this forest were cleared when the railway system came to the region in the 1920s. Crop plantations (e.g., coffee and cotton) and livestock raising expanded rapidly following deforestation as these activities attracted new settlers. A constantly growing population has put great pressure on the remaining forest fragments.

Our study area overlaps with the region formerly known as the 'Great Reserve of Pontal do Paranapanema', an area created in 1942 by the governor of the State of São Paulo (Ferrari-Leite 1998) to protect the native flora and fauna (Valladares-Pádua 1987). Pontal do Paranapanema is one of the most recently occupied areas of São Paulo State, with deforestation occurring later here than in other parts of the state. Until the 1950s, the region was extensively covered by semideciduous seasonal forests (297 000 ha). However, even after being given legally protected status in 1942, it continued to be reduced by illegal timber extraction and by land clearing for coffee and cotton plantations and pasturelands for cattle grazing.

In the 1950s, the state governor, Ademar de Barros, awarded land to his friends and political associates. Forest cover was further reduced as most of the land became consolidated into the hands of a few farmers, with 8% of the rural landowners now controlling 75% of the 246 840 ha of the Great Reserve of Pontal (ITESP 2002).

By the 1960s, Pontal had become one of the most degraded regions in São Paulo State. During the 1970s, the construction of dams and hydroelectric power plants created temporary employment and more settlers arrived from other parts of the country. People displaced from areas flooded by dams occupied new land, and sugar cane cultivation was initiated on a large scale.

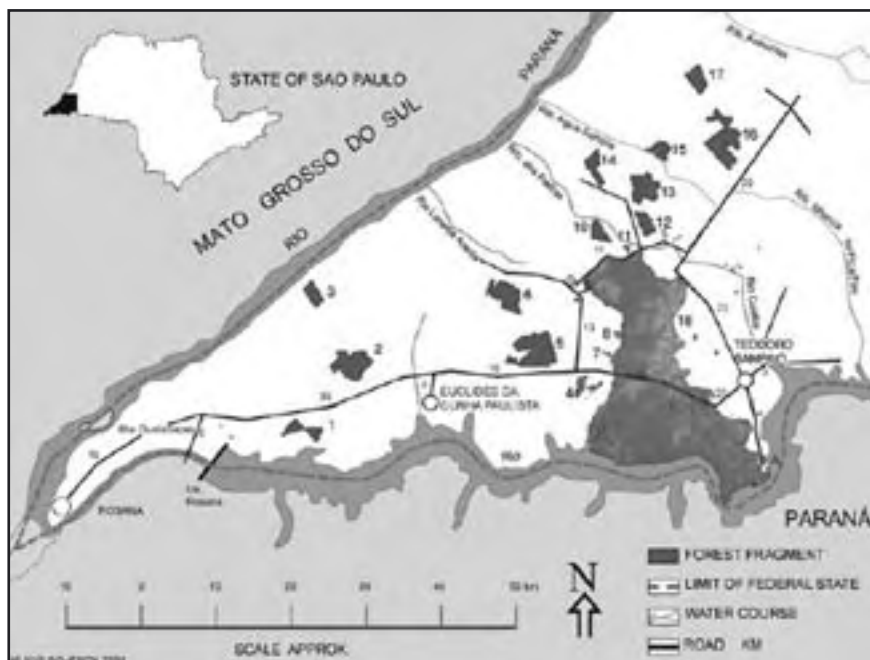


Figure 1. Map of the Pontal do Paranapanema region. The large shaded area is Morro do Diabo State Park. Fragments 5 and 9 are Tucano and Madre Cristina. The map shows only the larger forest fragments.

In the early 1990s, rural workers organized themselves into associations and started occupying and exploiting land in the region (Beduschi 2003). In the late 1990s, the Movimento dos Trabalhadores Rurais sem Terra – MST (Landless Movement of Rural Workers) began to contest the unfair distribution of land and encouraged ‘colonization’ of the region by landless workers. Settlements were established around the Morro do Diabo State Park (Figure 1) and in some of the forest fragments. This had a further negative effect on the few remaining forest fragments as people collected fuelwood, hunted and started fires to clear areas for cultivation (Cullen 1998).

Today Pontal is still a rather sparsely populated region occupied by large farms that were illegally established in past decades. Livestock raising is now the main activity in the predominantly agricultural economy of Pontal, and between 1996 and 2000 pastureland for grazing increased from 21 197 ha to 51 844 ha, resulting in the progressive loss of soil fertility. Official data reveal that pasturelands have now substantially replaced the forest cover and today account for about 60–70% of the Pontal region (ITESP 2002).

2.1 Current land use patterns in Pontal

Original Pontal forest cover was part of the Mata Atlântica (Atlantic Forest) biome (Joly *et al.* 1999), which was interspersed with patches of ‘cerrado’, a woodland-savannah vegetation representing the second largest biome in Brazil. Land use in the region currently consists of: (i) forested areas, principally in the Morro do Diabo State Park (MDSP) (35 000 ha) and also fragments in private and public hands (15 000 ha), totalling approximately 1.85% of the original forest; (ii) large farms owned by a few private individuals (240 000 ha); and (iii) settlements established by the government to relocate people immigrating to Pontal from other regions of the country (105 000 ha; Dean 1996).

In 2000, approximately 14 600 additional hectares were allocated to forest conservation, including Permanent Protection Areas and Legal Forest Reserves.

The major current threats to remaining forests are fire, hunting and illegal logging. Conservation measures need to be implemented as immigrants continue to come to the region and exert pressure on forest resources.

2.2 Effect of forest fragmentation

Habitat fragmentation is known to cause reduced plant population sizes and a consequent loss of alleles from the original (nonfragmented) gene pool (Frankel and Soulé 1981). Owing to their isolation and small size, remnant populations will continue to lose alleles through genetic drift (Ellstrand and Elam 1993). The genetic repercussions of habitat fragmentation – a loss of population genetic diversity – can be seen immediately after disturbance or over longer periods of time (Young *et al.* 1996). The reduction of natural habitat and the resulting spatial isolation of populations also have consequences for the reproductive success and gene flow of tree species (Templeton *et al.* 1990; Young *et al.* 1996; Nason and Hamrick 1997).

The long-term survival of fragmented populations of forest tree species depends on the implementation of adequate management strategies. To make these effective, it is first necessary to understand and to quantify the main impact of fragmentation on the patterns of distribution and the structure of genetic diversity. Such effects are more harmful to species that occur at lower densities, as these require more extensive areas to be preserved to secure sufficient numbers for regeneration and to avoid inbreeding. Such species can be used as models to infer the ecological and genetic effects that will be experienced by other species, and to help develop conservation measures to preserve the diversity and complexity of forest remnants. In order to explore a range of responses to isolation caused by fragmentation, for our study we chose forest tree species that had commercial value and were characterized by diverse biological traits.

3. Research

Research focused mainly on identifying the current effects of human activities on the genetic diversity of four tree species found in forest fragments and in the MDSP.

3.1 Study sites

The study sites were located in recent settlements, Madre Cristina and Tucano, that include forest fragments. A control site was established inside the MDSP.

Created in 1942, the MDSP covers an area of approximately 35 000 ha (Figure 2, overleaf). In the 1940s, clearings were opened inside the park by ‘posseiros’ (invading agricultural workers) who were later removed. The park is crossed by major roads and by a branch of the São Paulo State railway system, and in 1986 the São Paulo Power Company (CESP) cleared about 8% of the park while building the Rosana hydroelectric dam. Some selective logging of commercial species also occurred then. Nevertheless, large parts of the park are still conserved and represent a good sample of the original Pontal forest cover.

The Madre Cristina settlement was established in 1998. People had camped for three years along the side of the road before the settlement was officially recognized. Each of the 104 settled families now owns between 8 and 11 ha of land. Some families still live in flimsy shelters but the situation is improving. Sugar cane is the main crop, and settlers also collect wood for construction and fuel, and hunt in the forest fragment.

A road separates the MDSP from the main forest fragment of about 436 ha in Madre Cristina. This forest fragment is diverse in structure and species. The survey we carried out in 2000 revealed no obvious signs of recent logging, but an old extraction road cuts the fragment in two, and traces of selective logging from about 15 years ago were visible.

Established in 1991, Tucano is the oldest settlement in the area. With the arrival of settlers, the forest was cleared for crops and livestock raising. Plots for about 36

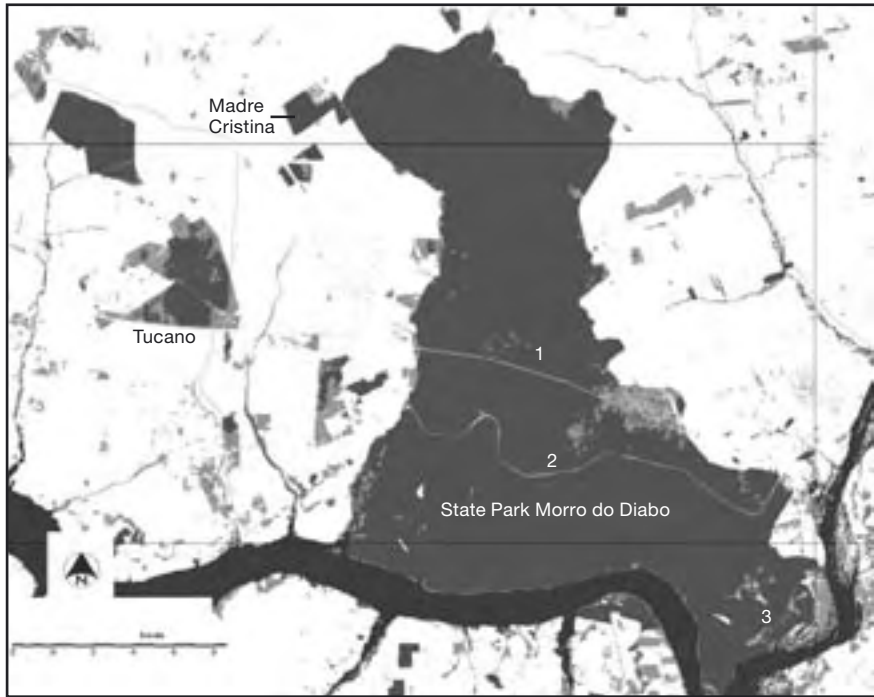


Figure 2. The study sites: Morro do Diabo State Park (MDSP) and the two forest fragments of Madre Cristina and Tucano. The asphalt state road (1) and the railway line (2) can be seen. MDSP headquarters are in the lower right-hand corner (3).

families average between 15 and 20 ha. Living conditions are relatively good, with access to electricity and clean water. Houses are of wood and are surrounded by fruit and ornamental trees. Corn, cotton and beans are cultivated, but cattle raising is the most important economic activity.

3.2 Ecological survey of selected species

Through consultations with settlers and nongovernmental organizations (NGOs), four species were selected for the study based on their ecological and economic importance. These were *Cedrela fissilis*, *Hymenaea courbaril*, *Peltophorum dubium*, and *Copaifera langsdorffii*. Table 1 (p. 172) summarizes relevant information about these species.

Surveys were undertaken in the three study areas (MDSP, Madre Cristina, and Tucano). For the four species at each study site, we mapped the distribution of individuals above 10 cm in diameter at breast height (dbh) and recorded their density within geo-referenced transects of 10 ha (20 x 5000 m). In order to maximize our chances of detecting the effects of fragmentation rather than the consequences of current forest resource use, transects were located in forest that seemed less-disturbed by human activity.

The four species were well represented at the three sites, and individuals were sampled from clusters of at least 30 contiguous adult individuals inside each plot. In addition, density of juveniles was studied in plots of 1600 m² immediately surrounding ten of the adult trees. The juveniles were grouped into three classes: class 1 (JI) with height between 0.5–1.0 m; class 2 (JII) with height between 1.0–2.0 m; and class 3 (JIII) with height above 2.0 m but not yet in the reproductive stage (see Figures 3a and 3b, on p. 176).

3.3 Socioeconomic study

We identified stakeholders and social actors at the three study sites. The rights, responsibilities and revenues of each stakeholder group and the relationships among them were then identified using the framework in the ‘Four Rs Methodology’ (Dubois and Lowore 2000). Information from the Madre Cristina and Tucano community members and leaders was obtained through informal interviews with five families and leaders of COCAMP (Cooperativa dos Assentados de Reforma Agrária do Pontal – Cooperative of Farmers Settled Through the Agrarian Reform in Pontal) and MST.

3.4 Forest genetic study

We undertook a forest genetic study aimed at understanding the correlation between the size of the forest fragments and the condition of their FGRs. We studied distribution of genetic diversity, inbreeding processes, current gene flow and genetic distance among populations from different fragments. The interpretation of the genetic data was supported by the availability of information on the use of nonwood forest products (NWFPs) and by data on the population size structure of the species investigated.

Sampling methods

Four exploratory transects were set up at each of the three study sites (MDSP, Madre Cristina and Tucano). Sub-plots, where at least 30 individuals above 30 cm dbh could be found for the four species, were identified in the transects and samples were collected. Population genetic structure was analyzed by sampling juveniles inside these subplots. Sample sizes are shown in Table 2.

Table 2. Number of adults and juveniles sampled of four species at three study sites in Brazil.

Species	Population	Adults	Juveniles	Total	
<i>Cedrela fissilis</i>	MDSP	30	27	57	
	Tucano	29	66	95	
	M. Cristina	28	22	50	202
<i>Hymenaea courbaril</i>	MDSP	30	29	59	
	Tucano	30	39	69	
	M. Cristina	27	24	51	179
<i>Copaifera langsdorffii</i>	MDSP	30	37	67	
	Tucano	30	34	64	
	M. Cristina	30	40	70	201
<i>Peltophorum dubium</i>	MDSP	30	72	102	
	Tucano	30	60	90	
	M. Cristina	28	40	68	260
Average		29.33	40.83		
Total		352	490		842

Table 1. Characteristics of *Peltophorum dubium*, *Cedrela fissilis*, *Hymenaea courbaril* and *Copaifera langsdorffii*.

Latin name	<i>Peltophorum dubium</i> Sprengel (<i>syns.</i> = <i>Caesalpinia dubia</i> Sprengel ^{1,2} ; <i>Peltophorum vogelianum</i> Benth ¹)	<i>Cedrela fissilis</i> Vellozo (<i>syns.</i> = <i>C. brasiliensis</i> Adr. Jussieu; <i>C. macrocarpa</i> Ducke; <i>C. tubiflora</i> Berton)	<i>Hymenaea courbaril</i> L. (<i>syns.</i> = <i>H. stilbocarpa</i> Hayne; <i>H. confertifolia</i> Hayne)	<i>Copaifera langsdorffii</i> Desf. (<i>syns.</i> = <i>C. grandifolia</i> [Benth ^{am}] Malme; <i>C. nitida</i> Hayne)
Common name	Canafistula	Cedro	Jatobá	Copaiba
Height [m] and dbh ¹ [cm]	Height: 10–20 ¹ ; 25–30 (40) ² dbh: 40–90 ¹ (300 ^{1,3,4}).	Height: 10–25 (40) dbh: 40–80 (200)	Height: 10–15 (25) dbh: 40–80 (120)	Height: 5–15 (35) dbh: 20–60 (100)
Successional group	Pioneer ^{4,5} to secondary initial ^{1,4}	Secondary initial to late secondary	Late secondary to climax	Late secondary to climax
Distribution	Very rare in climax and disturbed forest	In clusters of 1–3 individuals ⁴ Frequent? in disturbed forest	In clusters of about 4 individuals with short distances between individuals	Grouped in large clusters with great distances between clusters
Leaf-change	Semi-evergreen to deciduous ¹	Deciduous	Evergreen to semi-evergreen	Evergreen to semi-evergreen
Sex	Monoecious	Monoecious	Monoecious	Monoecious
Flowers	Bisexual (hermaphroditic) ^{1,6}	Functionally unisexual	Bisexual. Auto-incompatibility (post zygotic rejection)	Bisexual?
Flowering	December to March ¹	September to January (May) Irregular cycles. At the beginning of the wet season and immediately after the sprouting of leaves	October to December (February) Flowering irregular, once every 2–5 years (supra-annual, cycles can be different between individuals) Heavy flowering in 2000	(October to January) December to March Flowering and fruiting irregular (supra-annual)

Pollination: pollinators and time (day/night)	Insects. Nocturnal and/or diurnal?	Insects, possibly by moth and bees. Nocturnal and/or diurnal?	Principally bats. Hummingbirds and hymenoptera (bees). Mainly nocturnal	Insects. Bees (<i>Apis mellifera</i> and <i>Trigona</i> sp.), Bats (<i>Coptotermes curvignathus</i>)
Fruiting (mature fruits in São Paulo State)	May to December ¹	June to September. In some years trees do not produce fruits	June to December	August to October
Fruit [cm]	Plain and dry legume ¹ . Winged on both sides ⁶ . Length 4–9 ¹ ; 5–9 ^{2,6,7} ; Width 1–2.5 ¹ ; 1–1.7 ^{2,6} ; 1–1.5 ⁷ .	Ligneous capsule. Length 3–10 Width 3–3.5.	Ligneous pod, filled with edible fleshy pulp. Length 12–17 Width 5	Dry pod (ovoid). Length 4–5. Width 2–3
Seed	Long and hard ¹ . 1–3 seeds per fruit ^{1,2}	Winged on one side. About 30–100 seeds per fruit	Oval, wine colour. 2–8 seeds per fruit	Black (dark), long, partially covered with orange aril, fleshy. 1 seed per fruit
Seed size [mm]	Small. Length up to 10 ^{1,2,6} . Width 4 ^{1,6}	Large. Length up to 35. Width up to 15	Large. Length up to 20	Medium. Length up to 15. Width up to 8
Seed dispersion	Wind and self-dispersed ¹ . Seeds in seed bank in soil ¹	Wind dispersed	Animal dispersed: large mammals like Brazilian tapir (<i>Tapirus terrestris</i>), paca (<i>Cuniculus paca</i>), peccaries (<i>Tayassuidae</i>). Capuchin monkeys (<i>Cebus apella</i>) ⁸ , agoutis (<i>Dasyprocta punctata</i>) ¹⁰ , scaly-headed parrot (<i>Pionus maximiliani</i>) ⁹ . Gravity dispersed (barochory) ¹	Animal dispersed: birds such as tucanagu (<i>Ramphastos toco</i> , Rhamphastidae) and gralha-do- campo (<i>Cyanocorax cristatellus</i> , Corvidae) ¹¹ . Also water dispersed (frequent occurrence near water courses) ¹ . Gravity dispersed ¹
Seed bank	Yes ¹	No	Information not available	Information not available

Adapted from Ellenberg and Schenck 2001. ? = not known, + = diameter of tree stem at breast height, ++ = Personal communication of J. dos Santos USP/ESALQ, ¹ = Carvalho 1994, ² = Biloni 1990, ³ = Anonymous 1992, ⁴ = Sebbenn *et al.* 1999, ⁵ = Inoue and Galvão 1986, ⁶ = Orfila 1995, ⁷ = Burkart 1952, ⁸ = Galetti and Pedroni 1994, ⁹ = Hallwachs 1986, ¹⁰ = Galetti 1993, ¹¹ = Motta and Lombardi 1990. Figures in parentheses = maximum values recorded.

Laboratory methods

Leaf samples from *Hymenaea courbaril*, *Cedrela fissilis* and *Copaifera langsdorffii* were collected for DNA extraction, performed according to the CTAB (cetyltrimethylammonium bromide) protocol (Ferreira and Grattapaglia 1995). The selected SSR (simple sequence repeats or microsatellites) loci for each species and their sources are presented in Table 3. A protocol for allozyme markers in *Peltophorum dubium* was defined. Four gel/electrode buffer systems were tested and combined with 32 enzyme systems for leaf tissue samples from adults and juveniles.

Data analysis

Patterns of allelic diversity were examined in adults and juveniles at each site. The parameters estimated to evaluate the intrapopulation genetic structure were:

- Allelic richness or mean number of alleles per locus (A)
- Effective number (n_e) of alleles per locus ($n_e = 1/\sum p_i$ where p_i is the frequency of the i^{th} allele)
- Expected heterozygosity (H_e) as a measurement of genetic diversity
- Observed heterozygosity (H_o).

Estimates of genetic diversity within life stages and study sites were obtained by bootstrapping across loci, using the software GDA (Lewis and Zaykin 2000) and GenAlEx (Peakall and Smouse 2001). F-statistics were calculated according to the formula of Weir and Cockerham (1984) that measures genetic structure by partitioning variation through an analysis of variance. The parameters estimated were:

- Mean coefficient of inbreeding within populations (f)
- Total coefficient of inbreeding of the species (F)
- Co-ancestry coefficient between subpopulations or genetic drift between subpopulations (θ_p).

Table 3. Primers used for three species, number of alleles and allelic range.

Species	Primer	No. of alleles	Range	Source
<i>Cedrela fissilis</i>	CF 34	17	128–164	Markers developed by Gandara (unpublished data), at CENARGEN (Centro Nacional de Recursos Genéticos e Biotecnologia), EMBRAPA (Empresa Brasileira de Pesquisa Agropecuária)
	CF 66 A	21	118–154	
	CF 66 B	13	182–254	
	CF 78	19	114–168	
<i>Hymenaea courbaril</i>	HC 17	21	108–150	Ciampi 1999
	HC 25	6	118–144	
	HC 35	26	240–310	
	HC 40	19	162–196	
	HC 42	13	116–158	
	HC 48	25	130–180	
	HC 49	16	86–146	
<i>Copaifera langsdorffii</i>	CL 01	19	170–210	Ciampi 1999
	CL 02	19	252–310	
	CL 06	21	128–178	
	CL 20	21	92–136	
	CL 27	15	172–200	
	CL 34	15	182–208	

The confidence interval of 99% probability was estimated by bootstrapping across loci, using 10 000 replications. Gene flow (N_m) was calculated indirectly through estimates of F_{ST} (genetic diversity between populations) according to the following model:

$$N_m = (1 - F_{ST}) / 4 F_{ST}$$

The genetic distances between populations, for both adults and juveniles, were estimated by Nei's measure of genetic distance (Nei 1973) using the GDA software.

4. Results

4.1 Principal economic activities in the Pontal region

We identified stakeholders and then analyzed their relationships with the natural resources of the MDSP and the two study site forest fragments. Forty-two households were surveyed in the various settlements surrounding the MDSP to understand the role of NWFPs in their livelihoods. We found that settlers mostly collect NWFPs for internal consumption and that this activity did not contribute substantially to income. The settlers' principal monetary income activities are listed in Table 4. Livestock raising, practised on extensive pasturelands, was the most important source of income. Livestock grazing areas averaged between 3.5 and 7 hectares per farmer.

Several families in each settlement cultivated crops and fruit trees for internal consumption and/or commerce. While most agricultural production was not traded, cassava, maize, coffee, sugar cane and beans were sold in small quantities in local markets.

Table 4. Number of families involved in each economic activity.

Activity	TU	MC	SB	AS	Other	Total
Crop cultivation	1	0	2	1	1	5
Livestock	4	8	5	3	5	25
Crops + livestock	2	0	3	1	6	12

TU = Tucano settlement, MC = Madre Cristina settlement, SB = São Bento settlement, AS = Água Sumida settlement, Other = set of settlements: Paulo Freire, Antonio Conselheiro, Che Guevara and Bom Pastor.

4.2 Population size and structure of juveniles of the four selected species

Analysis of the population structure of the four species revealed differences across species and plots (Figures 3a and 3b, overleaf), although no clear patterns emerged.

Our surveys showed that there were still large populations of all four species at the three sites, indicating that in the near term these species are not threatened by fragmentation. In fact, regeneration of *Hymenaea courbaril* and *Cedrela fissilis* seemed to be greater in the forest fragments than in the MDSP, though the significance of this must be tempered by the fact that fragmentation has taken place recently.

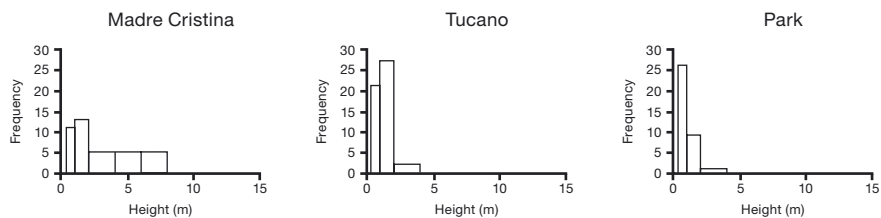
4.3 Forest genetic study

Cedrela fissilis

Data from 202 individuals from three populations of *Cedrela fissilis* (87 adults and 115 juveniles) were analyzed (Table 5, on p. 177).

Even though a significant loss of alleles was not detected in *C. fissilis*, inbreeding was found, and this could lead to loss of alleles in future generations. Indeed, juveniles show much greater inbreeding than adults, as indicated by the higher value of N_m (see Table 6, on p. 177).

Peltophorum dubium



Cedrela fissilis

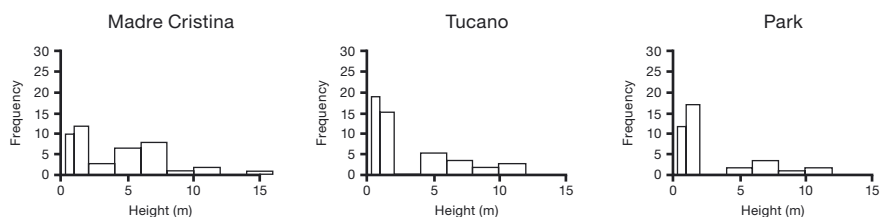
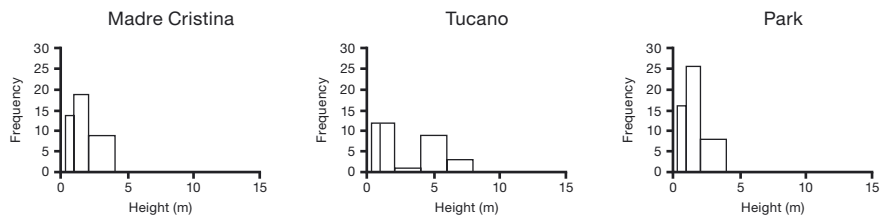


Figure 3a. 2003 survey of juvenile trees (JI, JII and JIII) of *Peltophorum dubium* and *Cedrela fissilis* in Morro do Diabo State Park and Tucano and Madre Cristina forest fragments.

Copaifera langsdorffii



Hymenaea courbaril

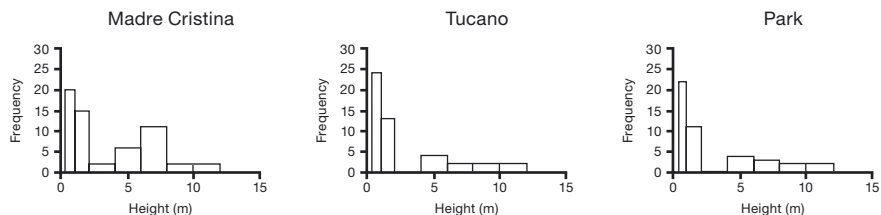


Figure 3b. 2003 survey of juvenile trees (JI, JII and JIII) of *Copaifera langsdorffii* and *Hymenaea courbaril* in Morro do Diabo State Park and Tucano and Madre Cristina forest fragments.

Table 5. Genetic parameters of three *C. fissilis* populations (MDSP, Tucano and Madre Cristina settlements).

Adults	n	A	n_e	H_e	H_o
MDSP	29.50 (0.50)	12.50 (3.20)	6.41 (2.66)	0.80 (0.11)	0.75 (0.15)
Madre Cristina	25.00 (1.41)	10.50 (3.35)	5.86 (2.51)	0.80 (0.08)	0.66 (0.16)
Tucano	27.00 (1.58)	12.75 (3.56)	7.48 (2.45)	0.85 (0.06)	0.66 (0.28)
Mean	27.17 (2.25)	11.91 (1.23)	6.58 (0.82)	0.82 (0.03)	0.69 (0.05)
Juveniles	n	A	n_e	H_e	H_o
MDSP	21.00 (5.96)	10.75 (3.56)	5.67 (2.36)	0.76 (0.16)	0.72 (0.27)
Madre Cristina	19.25 (1.78)	9.75 (3.11)	6.00 (2.50)	0.77 (0.16)	0.70 (0.16)
Tucano	58.75 (3.27)	17.00 (1.22)	7.73 (2.05)	0.86 (0.04)	0.61 (0.14)
Mean	33.00 (22.32)	12.50 (3.93)	6.47 (1.10)	0.80 (0.05)	0.68 (0.06)
Overall average	30.08 (14.54)	12.20 (2.62)	6.52 (0.87)	0.80 (0.04)	0.68 (0.05)

The parameters obtained for juveniles and adults were number of genotyped individuals (n), average number of alleles per locus (A), effective number of alleles per locus (n_e), expected heterozygosity (H_e) and observed heterozygosity (H_o). Values in parentheses indicate the standard deviation of the estimates.

Nevertheless, forest fragmentation does not appear to have affected gene flow much, perhaps because of the mobile character of pollinators (bees) and the fairly close spatial distribution of fragments in the landscape.

Analysis of the population structure (Table 6) of *C. fissilis* reveals that inbreeding in adult trees results from contributions of both the mating system ($f = 0.163$) and the fragmentation of populations ($\theta_p = 0.067$). Comparing the likely age of the adult individuals sampled with the time when disturbance and fragmentation began in Pontal, it seems that inbreeding was already occurring before the isolation of fragments took place. The Tucano fragment is an exception, as the forest has been affected not only by fragmentation but also by reduction in the density of individuals by selective logging in the more recent past. The indirect estimate of N_m indicated that gene flow was slowing the effects of genetic drift in the populations sampled, although some genetic differentiation was found. Estimates

Table 6. Parameters of genetic structure obtained for three adult and three juvenile populations of *C. fissilis*. Confidence interval of 99% was obtained through 10 000 bootstraps over loci.

Parameter	Adults		Juveniles	
	Estimated value	Confidence interval	Estimated value	Confidence interval
f	0.163	0.059–0.371	0.217	0.110–0.380
F	0.219	0.082–0.464	0.242	0.142–0.409
θ_p	0.067	0.023–0.148	0.032	0.016–0.052
N_m	3.487		7.466	

of the genetic structure in juveniles suggested a smaller differentiation than among the adults, and there appeared to be higher gene flow.

The main pollinators of this species are bees, and seeds are wind-dispersed. Thus, the matrix of open pastureland between fragments may be facilitating more gene flow than would normally occur in wind-pollinated tree species (Young *et al.* 1993).

Gene flow differed across forest fragments as indicated by Nei genetic distance.

The juvenile and adult populations in the MDSP were genetically distant from the populations in Tucano and Madre Cristina, with the greatest genetic distance found between the adult populations in the park and Madre Cristina (Table 7).

Table 7. Matrices of Nei genetic distances among the adult populations of *C. fissilis*.

Adults	MDSP	Tucano	Madre Cristina
MDSP	-	-	-
Tucano	0.3949	-	-
Madre Cristina	0.6671	0.3003	-
Juveniles	MDSP	Tucano	Madre Cristina
MDSP	-	-	-
Tucano	0.2618	-	-
Madre Cristina	0.0987	0.1052	-

Hymenaea courbaril

One hundred and eighty individuals of *Hymenaea courbaril* were sampled (87 adults and 93 juveniles) from the three populations (MDSP, Tucano, Madre Cristina). The average number of alleles per locus (A) was higher in adults than in juveniles (Table 8), but the difference was not statistically significant.

Estimates of genetic diversity were similar in the three adult tree populations, but considerable differences in observed heterozygosity were detected (Table 8). In the MDSP, adults showed a greater value of observed heterozygosity than expected heterozygosity, revealing high outcrossing rates. In Tucano, adults showed a high level of inbreeding, probably owing to the selective logging that occurred after fragmentation.

The level of inbreeding was on average higher in juveniles than in adults at all three sites. However, fragmentation did not seem to be depressing gene flow in the three populations or causing alterations in genetic structure, as the value of gene flow in juveniles and adults is not significantly different (Table 9). This could be linked to the abundance of the main agents of dispersal (rodents and large, hoofed animals) that are still found in good numbers in the study areas. The regeneration of this species seems to be favoured by a forest edge effect, which implies more disturbed conditions, different quality and greater intensity of light, and a different microclimate from the forest interior.

The Nei genetic distances between the populations in Madre Cristina and Tucano were higher in juveniles than in adults (Table 10). This suggests more inbreeding in the fragments owing to the more pronounced isolation of individuals. The genetic distances show that gene flow did not occur with similar intensities in the three populations. The greatest distance was found between adults in the MDSP and in the Madre Cristina fragment.

Table 8. Genetic parameters of three *H. courbaril* populations (MDSP, Tucano and Madre Cristina settlements).

Adults	n	A	n_e	H_e	H_o
MDSP	28.00 (3.41)	9.43 (4.65)	5.67 (2.56)	0.77 (0.15)	0.86 (0.19)
Madre Cristina	26.00 (1.53)	8.43 (3.35)	5.13 (2.08)	0.78 (0.07)	0.60 (0.17)
Tucano	27.29 (3.45)	9.14 (4.56)	5.72 (3.76)	0.73 (0.18)	0.39 (0.22)
Mean	27.10 (1.01)	9.00 (0.51)	5.50 (0.32)	0.76 (0.26)	0.62 (0.23)
Juveniles	n	A	n_e	H_e	H_o
MDSP	26.57 (3.82)	7.43 (2.76)	4.09 (1.40)	0.72 (0.11)	0.44 (0.11)
Madre Cristina	21.57 (2.82)	7.14 (2.61)	3.91 (1.64)	0.71 (0.10)	0.55 (0.30)
Tucano	36.14 (2.61)	8.00 (3.56)	4.91 (3.24)	0.73 (0.13)	0.55 (0.11)
Mean	28.09 (7.40)	7.52 (0.44)	4.30 (0.53)	0.72 (0.01)	0.51 (0.06)
Overall average	27.59 (4.76)	8.26 (0.91)	4.90 (0.77)	0.74 (0.03)	0.56 (0.16)

The parameters obtained for juveniles and adults were number of genotyped individuals (n), average number of alleles per locus (A), effective number of alleles per locus (n_e), expected heterozygosity (H_e) and observed heterozygosity (H_o). Values in parentheses indicate the standard deviation of the estimates.

Table 9. Parameters of genetic structure obtained for three adult and three juvenile populations of *H. courbaril*. Confidence interval of 99% was obtained through 10 000 bootstraps over loci.

Parameter	Adults		Juveniles	
	Estimated value	Confidence interval	Estimated value	Confidence interval
f	0.207	0.047–0.338	0.284	0.156–0.437
F	0.288	0.137–0.403	0.355	0.231–0.498
θ_p	0.102	0.059–0.149	0.099	0.065–0.160
N_m	2.199		2.262	

Table 10. Matrices of Nei genetic distances among the adult and juvenile populations of *H. courbaril*.

Adults	MDSP	Tucano	Madre Cristina
MDSP	-	-	-
Tucano	0.5351	-	-
Madre Cristina	0.8300	0.1753	-
Juveniles	MDSP	Tucano	Madre Cristina
MDSP	-	-	-
Tucano	0.3645	-	-
Madre Cristina	0.4677	0.3876	-

Copaifera langsdorffii

Data from 201 copaiba trees were analyzed (90 adults and 111 juveniles). Like *Hymenaea courbaril* and *Cedrela fissilis*, the three populations of *C. langsdorffii* showed high levels of genetic diversity (Table 11).

Table 11. Genetic parameters of three *C. langsdorffii* populations (MDSP, Tucano and Madre Cristina settlements).

Adults	n	A	n_e	H_e	H_o
MDSP	28.67 (0.82)	10.50 (2.59)	5.02 (0.98)	0.80 (0.03)	0.58 (0.21)
Madre Cristina	25.83 (3.37)	9.83 (2.86)	4.84 (2.04)	0.77 (0.07)	0.47 (0.21)
Tucano	28.50 (2.07)	10.67 (2.34)	5.76 (1.37)	0.82 (0.04)	0.53 (0.13)
Mean	27.67 (1.59)	10.33 (0.44)	5.21 (0.49)	0.80 (0.02)	0.53 (0.05)
Juveniles	n	A	n_e	H_e	H_o
MDSP	29.33 (1.37)	11.00 (2.37)	5.28 (2.46)	0.78 (0.09)	0.55 (0.24)
Madre Cristina	37.83 (1.17)	12.00 (3.29)	5.57 (2.08)	0.80 (0.08)	0.58 (0.23)
Tucano	27.83 (4.58)	9.17 (4.35)	4.69 (3.17)	0.73 (0.11)	0.59 (0.26)
Mean	31.66 (5.39)	10.72 (1.43)	5.18 (0.45)	0.77 (0.04)	0.57 (0.02)
Overall average	29.66 (4.17)	10.52 (0.97)	5.19 (0.42)	0.78 (0.03)	0.55 (0.04)

The parameters obtained for juveniles and adults were number of genotyped individuals (n), average number of alleles per locus (A), effective number of alleles per locus (n_e), expected heterozygosity (H_e) and observed heterozygosity (H_o). Values in parentheses indicate the standard deviation of the estimates.

The average number of alleles per locus (A) and genetic diversity were high in the adult trees of the three populations. The effective number of alleles per locus (n_e) was much lower than allelic richness (A), indicating the presence of alleles with low frequency, and therefore suggesting that genetic drift was not strong. However, the fixation index (f) was high in the three populations. As with *Cedrela fissilis* and *Hymenaea courbaril*, the population of *C. langsdorffii* inside the MDSP presented the smallest indices of inbreeding (Table 12).

The analysis of genetic diversity among *C. langsdorffii* populations showed that most of the existing allelic variability is within populations (Table 12). The main contributor to total inbreeding ($F = 0.360$) seems to be the mating system of this species ($f = 0.341$) and not fragmentation. The lower value of f versus F , the low estimate of θ_s , and the estimate of apparent gene flow all seem to suggest that populations from the three sites were not undergoing a significant differentiation process. The parameters of genetic structure for *C. langsdorffii* match those expected based on its reproductive biology; the species is distributed in small clusters at low densities, with pollen travelling long distances by bees. Seed dispersal is by birds that, in general, regurgitate seeds near the mother plant.

The estimates of the genetic structure obtained for the juveniles indicate smaller indices of inbreeding in this class compared to the adults (Table 12). However, when observing the confidence intervals, differences in inbreeding indices between the juveniles and the adults are subtle.

Forest fragmentation has resulted in restricted gene flow among the *C. langsdorffii* populations. The species' infrequent and partly asynchronous blooming (Pedroni *et al.* 2002) can amplify the effects of fragmentation. Even though there was a lot of inbreeding,

Table 12. Parameters of genetic structure for three adult and three juvenile populations of *C. langsdorffii*. Confidence interval of 99% was obtained by 10 000 bootstraps on loci.

Parameter	Adults		Juveniles	
	Estimated value	Confidence interval	Estimated value	Confidence interval
f	0.341	0.220–0.512	0.270	0.108–0.416
F	0.360	0.240–0.530	0.320	0.164–0.467
θ_p	0.029	0.014–0.042	0.069	0.038–0.099
N_m	8.491		3.352	

genetic diversity in populations with different blooming periods was high. This phenomenon has also been observed in the tropical tree species *Pithecellobium elegans* (Hall *et al.* 1996).

Genetic distances between the adult populations were smaller than the other species for all the possible pairs (Table 13), indicating that gene flow occurred in similar intensities among the three populations and that genetic drift and selection did not cause differentiation among the populations. The data also indicate that gene flow was restricted mostly between populations located furthest away from each other (MDSP and the Tucano fragment).

Peltophorum dubium

The adult individuals of *Peltophorum dubium* showed high genetic diversity in the three forest sites (Table 14, overleaf). The average number of alleles per locus (A) and the values of heterozygosity (H_e and H_o) were lower than the ones obtained for the other three species. This is due to the lower capacity of allozyme electrophoresis (with co-dominant markers) to detect polymorphisms when compared to microsatellites.

Genetic diversity was high for *P. dubium* juveniles ($H_e = 0.463$) and adults ($H_e = 0.451$), with no differences detected among the three populations for either H_o or H_e . The populations studied presented a practically null fixation index.

In contrast to what was observed in the other species, the estimate of total inbreeding in *P. dubium* did not differ significantly from zero (Table 15, overleaf). This probably

Table 13. Matrices of Nei genetic distances among the adult and juvenile populations of *C. langsdorffii*.

Adults	MDSP	Tucano	Madre Cristina
MDSP	-	-	-
Tucano	0.1445	-	-
Madre Cristina	0.1268	0.1960	-
Juveniles	MDSP	Tucano	Madre Cristina
MDSP	-	-	-
Tucano	0.5670	-	-
Madre Cristina	0.1613	0.3319	-

Table 14. Genetic parameters of three *P. dubium* populations (MDSP, Tucano and Madre Cristina settlements).

Adults	n	A	n _e	H _e	H _o
MDSP	26.20 (1.24)	2.40 (0.00)	1.85 (0.60)	0.40 (0.23)	0.41 (0.25)
Madre Cristina	22.40 (3.12)	2.40 (0.00)	1.99 (0.69)	0.45 (0.17)	0.48 (0.19)
Tucano	24.20 (1.61)	2.40 (0.00)	2.08 (0.72)	0.47 (0.18)	0.46 (0.17)
Mean	24.27 (1.90)	2.40 (0.00)	1.97 (0.12)	0.45 (0.04)	0.45 (0.04)
Juveniles	n	A	n _e	H _e	H _o
MDSP	19.93 (0.50)	2.40 (0.00)	2.07 (0.76)	0.46 (0.20)	0.40 (0.14)
Madre Cristina	10.07 (5.80)	2.20 (0.20)	1.89 (0.42)	0.45 (0.12)	0.49 (0.09)
Tucano	13.87 (1.60)	2.40 (0.00)	2.01 (0.51)	0.48 (0.12)	0.48 (0.16)
Mean	14.62 (5.26)	2.33 (0.14)	1.99 (0.09)	0.46 (0.04)	0.45 (0.09)
Overall average	17.03 (6.31)	2.35 (0.12)	1.97 (0.12)	0.46 (0.04)	0.45 (0.08)

The parameters obtained for juveniles and adults were number of genotyped individuals (n), average number of alleles per locus (A), effective number of alleles per locus (n_e), expected heterozygosity (H_e) and observed heterozygosity (H_o). Values in parentheses indicate the standard deviation of the estimates.

Table 15. Estimates of genetic structure obtained as averages on loci for adult and juvenile populations of *P. dubium*. Confidence interval of 95% was estimated from 10 000 bootstraps over loci.

Parameters	Adults		Juveniles	
	Estimated value	Confidence interval	Estimated value	Confidence interval
<i>f</i>	0.0012	-0.2030–0.1839	0.0396	-0.1471–0.1733
<i>F</i>	0.0279	-0.1531–0.2082	0.0523	-0.1490–0.1879
θ_p	0.0267	0.0022–0.0578	-0.0019	-0.0155–0.0092
<i>N_m</i>	9.11		18.94	

indicates that *P. dubium* has developed an efficient pollination system typical of species that reproduce using cross fertilization. A low divergence among the populations was observed ($\theta_p = 0.027$) and a balance between gene flow and genetic drift was found. Genetic divergence among juveniles was close to zero (Table 15).

5. Concluding remarks

The land use changes that have occurred in Pontal since the 1950s have led to a near-complete removal of the natural forest, which has now been replaced mostly by pastureland. Livestock raising is currently the most important economic activity in the southwestern part of São Paulo State. Not only did the forest cover suffer major reductions in the recent

past, the few fragments that remain continue to be threatened by illegal logging, fuelwood collection and disturbance linked to hunting.

The genetic diversity research that we undertook indicated that forest fragmentation has not yet resulted in a significant loss of alleles in the adult populations of the four selected species. In fact, although two of the populations studied were located in fragments, these were large fragments and the loss of adult trees was not so drastic as to cause immediate and significant loss of genetic diversity; the genetic diversity also remained high in juveniles. Indeed, the populations of all four species displayed high levels of genetic diversity, contrary to what has been observed for rare forest species elsewhere (Hamrick and Murawski 1991). However, it must be emphasized that only recently have microsatellite markers been used to study tropical arboreal species. Therefore, the genetic parameter estimates obtained here were probably higher than those that would have been obtained with other methods because of their higher capacity to detect polymorphisms.

Because of fragmentation, we expected to find spatial structure in the patterns of genes. However, we observed little spatial organization in adult populations of *Cedrela fissilis*, *Peltophorum dubium* and *Copaifera langsdorffii* (while a more pronounced genetic distance between populations was found in *Hymenaea courbaril*). The low genetic differentiation found among the populations examined in this study is consistent with the high outcrossing rates that tropical tree species usually exhibit (Hall *et al.* 1996). Considering the differences in the ecological and reproductive characteristics of the species studied, outcrossing rates by themselves are not enough to predict the degree and the type of differentiation noted in these populations. For example, species that occur naturally in meta-population structures will probably not have their genetic structure modified very much after fragmentation events (England *et al.* 2002).

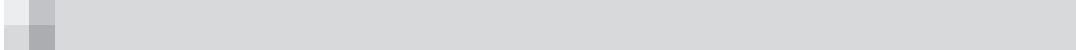
Nevertheless, environmental awareness is increasing in the Pontal region, and forest planting and other recovery programmes are beginning to be successful. Legislation and policies are increasingly directed towards conservation of forest remnants by restricting exploitation of both timber and NWFPs. Enforcement of the existing laws is also improving, albeit slowly. In addition, farmers' organizations like COCAMP and MST have incorporated environmental considerations into their principles and practices. Some farmers are starting to adopt agroforestry programmes using native species in areas bordering forest fragments, including the adoption of silvipastoral systems.

References

- Anonymous. 1992. Árboles de Misiones: *Peltophorum dubium*. Yvyrareta 3(3):25-27.
- Beduschi Filho, L.C. 2003. Assentamentos Rurais e Conservação da Natureza: Do Estranhamento à Ação Coletiva. Iglu/FAPESP (Fundação de Amparo à Pesquisa do Estado de São Paulo), São Paulo, Brazil.
- Biloni, J.S. 1990. Árboles autóctonos Argentinos. Typography Argentina Publishing company, Buenos Aires, Argentina.
- Burkart, A. 1952. Las Leguminosas Argentinas. Editorial Acme Agency, Buenos Aires, Argentina.
- Carvalho, P.E.R. 1994. Espécies florestais brasileiras: recomendações silviculturais, potencialidades e uso de madeira. Brasília. EMBRAPA-CNPQ (Empresa Brasileira de Pesquisa Agropecuária-Centro Nacional de Pesquisa Florestal). Brasília, Argentina.
- Ciampi, A.Y. 1999. Desenvolvimento e utilização de marcadores microsatélites, AFLP e sequenciamento de cpDNA, no estudo da estrutura genética e parentesco em populações de copaíba (*Copaifera langsdorffii*) em matas de galeria no cerrado. Botucatu. PhD thesis, Instituto de Biociências, Universidade Estadual Paulista Júlio de Mesquita Filho, Botucatu, Brazil.
- Cullen Jr, L. 1998. Ilhas de Biodiversidade como corredores na restauração de paisagens fragmentadas no Pontal do Paranapanema, São Paulo, Uma proposta para o Programa Nacional de Diversidade Biológica – PRONABIO. IPÊ – Instituto de Pesquisas Ecológicas, Nazaré Paulista, São Paulo, Brazil.

- Dean, W.A. 1996. A Ferro e Fogo: A História e a Devastação da Mata Atlântica Brasileira. Companhia das Letras, São Paulo, Brazil.
- Dubois, O. and J. Lowore. 2000. The journey towards collaborative forest management in Africa: Lessons learned and some navigational aids. An overview paper prepared for the Food and Agriculture Organization of the United Nations (FAO), International Institute for Environment and Development (IIED), London, UK.
- Ellenberg, H. and K. Schenck. 2001. Final report for the ecological research work in the project Conservation, Management and Sustainable Use of Forest Genetic Resources with Reference to Brazil and Argentina, to the International Plant Genetic Resources Institute (IPGRI), for the period May 2001–Sept 2002. IPGRI, Rome, Italy.
- Ellstrand, N.C. and D.R. Elam. 1993. Population genetic consequences of small population size: implications for plant conservation. *Annu. Rev. Ecol. Sys.* 24:217-242.
- England, P.R., A.V. Usher, R.J. Whelan and D.J. Ayre. 2002. Microsatellite diversity and genetic structure of fragmented populations of the rare, fire-dependent shrub *Grevillea macleayana*. *Mol. Ecol.* 11:976-977.
- Ferrari-Leite, J. 1998. A Ocupação do Pontal do Paranapanema. Editora Hucitec, São Paulo, Brazil.
- Ferreira, M.E. and D. Grattapaglia. 1995. Introdução ao uso de marcadores genéticos (2nd edn.) EMBRAPA-CENARGEN (Empresa Brasileira de Pesquisa Agropecuária-Centro Nacional de Recursos Genéticos e Biotecnologia), Brazil.
- Frankel, O.H. and M.E. Soulé. 1981. Conservation and Evolution. Cambridge University Press, Cambridge, UK.
- Galetti, M. 1993. Diet of the scaly-headed parrot (*Pionus maximiliani*) in a semideciduous forest in southeastern Brazil. *Biotropica* 25:419-425.
- Galetti, M. and F. Pedroni. 1994. Seasonal diet of capuchin monkeys (*Cebus apella*) in a semideciduous forest in southeast Brazil. *J. Trop. Ecol.* 10:27-38.
- Hall, P., S. Walker and K.S. Bawa. 1996. Effect of forest fragmentation on genetic diversity and mating system in a tropical tree, *Pithecellobium elegans*. *Conserv. Biol.* 10:757-768.
- Hallwachs, W. 1986. Agoutis (*Dasyprocta punctata*): The inheritors of Guapinol (*Hymenaea courbaril*, Leguminosae). In *Frugivores and Seed Dispersal* (A. Estrada and T.H. Fleming, eds.). Junk Publishers, Dordrecht, The Netherlands.
- Hamrick, J.L. and D.A. Murawski. 1991. Levels of allozyme diversity in populations of uncommon tropical tree species. *J. Trop. Ecol.* 7:395-399.
- Inoue, M.T. and F. Galvão. 1986. Desempenho assimilatório de *Mimosa scabrella*, *Peltophorum dubium*, *Schinus terebinthifolius* e *Matayba elaeagnoides*, em dependência da intensidade luminosa. *Acta Forestalia Brasiliensis*, Curitiba 1(1):89-98.
- ITESP (Instituto de Terras do Estado de São Paulo). 2002. Avaliação da Produção Agropecuária e da População dos Assentamentos do Estado de São Paulo no período 1996–2000. Departamento de Agricultura, Produção e Defesa, São Paulo, Brazil.
- Joly, C.A., M.P.M. Aidar, C.A. Klink, D.G. McGrath, A.G. Moreira, P. Moutinho, D.C. Nepstad, A.A. Oliveira, A. Pott, M.J.N. Rodal and E.V.S.B. Sampaio. 1999. Evolution of the Brazilian phytogeography classification systems: implications for biodiversity conservation. *Ciência e Cultura* 51:331-348.
- Lewis, P.O. and D. Zaykin. 2000. Genetic Data Analysis: Computer program for the analysis of allelic data. Version 1.0 (d15). Free program distributed by the authors over the Internet from the GDA. homepage at <http://hydrodictyon.eeb.uconn.edu/people/plewis/software.php>
- Motta Jr, J.C. and J.A. Lombardi. 1990. Aves como agentes dispersores de copaíba (*Copaifera langsdorffii*, Caesalpinaceae) em São Carlos, São Paulo State. *Ararajuba* 1:105-106.
- Nason, J.D. and J.L. Hamrick. 1997. Reproductive and genetic consequences of forest fragmentation: two case studies of neotropical canopy trees. *J. Hered.* 88:264-276.
- Nei, M. 1973. Analysis of gene diversity in subdivided populations. *Proc. Nat. Acad. Sci. USA* 70:3321-3323.
- Orfila, E.N. 1995. Frutas, Semillas y Plantulas de la Flora Argentina. Ediciones Sur, Buenos Aires, Argentina.

- Peakall, R. and P.E. Smouse. 2001. GenAIEx V5: Genetic Analysis in Excel. Population genetic software for teaching and research. Australian National University, Canberra, Australia. <http://www.anu.edu.au/BoZo/GenAIEx>
- Pedroni, F., M. Sanchez and F.A.M. Santos. 2002. Fenologia da copaíba (*Copaifera langsdorffii* Desf. – Leguminose, Caesalpinoideae) em uma floresta semidecídua no sudeste do Brasil. *Revista Brasileira de Botânica* 25(2):183-194.
- Sebbenn, A.M., A.C.M.F. Siqueira, R. Vencovsky and J.A.R. Machado. 1999. Interação genótipo ambiente na conservação ex situ de *Peltophorum dubium*, em duas regiões do Estado de São Paulo. *Rev. Inst. Flor, SP*, 11(1):75-89.
- Templeton, A.R., K. Shaw, E. Routman, S.K. Davis. 1990. The genetic consequences of habitat fragmentation. *Mo. Bot. Gard. Annu. Rep.* 77:13–27.
- Valladares-Pádua, C. 1987. Black lion tamarin (*Leontopithecus chrysopygus*): Status and conservation. MSc thesis, University of Florida, Gainesville, USA.
- Weir, B.S. and C.C. Cockerham. 1984. Estimating F-statistics for the analysis of population structure. *Evolution* 38:1358-1370.
- Young, A., T. Boyle and T. Brown. 1996. The population genetic consequences of habitat fragmentation for plants. *Trends Ecol. Evol.* 11:413-418.
- Young, A.G., H.G. Merriam and I. Warwick, 1993. The effects of forest fragmentation on genetic variation in *Acer saccharum* Marsh. (sugar maple) populations. *Heredity* 71:277-289.



Chapter 10

A modelling case study: options for FGR management in *Araucaria araucana* ecosystems

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1. Introduction

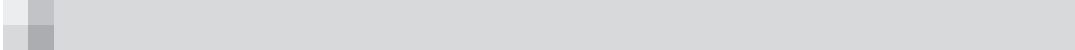
One of the major objectives of this forest genetic resource (FGR) project, funded by the German Federal Ministry for Economic Cooperation and Development (BMZ), was to develop and apply a general model that could improve our understanding of the dynamics of FGRs in forests used by local communities. Our ultimate goal was to use this model to compare the behaviour of different forest ecosystems, and ultimately to identify the factors that influence their genetic processes. We believed that this approach would help us to determine and then to monitor sustainable levels of FGRs for both management and conservation purposes. In this chapter, we present the results of our modelling activities for the *Araucaria araucana* (pehuén or araucaria) forests of Argentina.

2. The Pehuén Model

In the last 30 years or so, the field of forest conservation genetics has mostly focused on identifying centres of tree species' genetic diversity and investigating altered genetic processes that somehow jeopardized FGR conservation (Young *et al.* 2000). But in terms of real world applicability, this work has largely missed a major component of FGR conservation and use by not taking into account the most important actors in conservation: human beings.

Combined with such natural disturbances as fire, volcanism and windstorm, human actions have also shaped the characteristics of *Araucaria araucana* forests over time. As seen in Chapter 6, deforestation has significantly reduced the extension of Argentine araucaria forests (Veblen *et al.* 1999; Rechene 2000). Such major disturbances have affected the stability of araucaria ecosystems and the survival of their species. They have modified forest habitats and changed their floristic composition and genetic characteristics, but with an intensity that varies across the natural range of the species. Araucaria forests continue to change under the influence of human use. Selective logging and seed (piñones) collecting are now strongly affecting araucaria ecosystems and genetic processes, and this is especially true in the fragmented eastern populations of the species that now regenerate mostly by vegetative means. Therefore, when studying pehuén genetics and forest dynamics, the addition of the human element and its various socioeconomic contexts is essential, even though it increases the investigative and analytical challenge.

The complexity of such human-modified systems is not easily disentangled through linear thinking or strictly deterministic analytical tools. The dynamic interdependencies among the many parameters and variables to be considered when modelling such systems mean that a holistic approach needs to be used (Haraldsson 2000). Systems thinking and one of its components, system dynamics, enhances our ability to understand



dynamic interrelationships (Richmond 1993), and these are therefore appropriate tools for conceptualizing and analyzing entities like forests that we hope to conserve while still exploiting their resources (= 'conservation through use' systems).

Through a model that applies systems research approaches, we try to describe the pehuén forests of Argentina that are dominated by *Araucaria araucana* and inhabited by indigenous Mapuche people. The most important elements integrated into our model are:

- The extent of the forest ecosystem to be modelled and changes to it that result from human actions (= boundaries of the system)
- Seed productivity in different forests
- Seed consumption by domesticated and wild animals
- Seed collecting by Mapuche
- Forest genetic diversity
- Condition of pasturelands (productivity and carrying capacity)
- Amount and distribution of wealth within Mapuche communities, and
- Per capita Mapuche income from off-farm sources.

Throughout this chapter we call our modelled araucaria forest the Pehuén Model. We created it with two objectives in mind:

- To represent the structure and dynamics of araucaria forests through a descriptive, conceptual model that identifies links and feedbacks among genetic, ecological and socioeconomic processes and variables, and
- To predict the future behaviour of the system by factoring into the model both current management practices and more sustainable alternatives, thereby allowing us to evaluate the outcomes of different management regimes over time.

2.1 Materials and methods

The key species in the forest ecosystem we have modelled is *Araucaria araucana*. This large tree is native to Argentina and Chile, growing mostly at elevations of between 900 and 1800 metres above sea level (m asl), with sporadic occurrences at altitudes as low as 600 m asl (Armesto *et al.* 1997). *Araucaria* grows mostly on soils derived from recent volcanic ash deposits, though it can also be found on deeper soils derived from metamorphic and sedimentary rocks (Armesto *et al.* 1997). The natural distribution of araucaria covers a rainfall gradient ranging from 900 to 2500 mm of rainfall per year in Argentina, with some Chilean sites receiving up to 4000 mm per year.

Araucaria araucana mixes with other tree species along rainfall and latitudinal gradients, but it is most commonly associated with *Nothofagus* spp. and *Austrocedrus chilensis*. The three most common forest associations are:

- *Araucaria araucana* and *Nothofagus pumilio* (called 'lenga' in the Mapuche language). This association generally occurs in the western, wetter part of the araucaria range at higher elevations (1100–1800 m asl) on south-facing slopes. This forest type has been commercially exploited
- *Araucaria araucana* and *Nothofagus antarctica* in shrub form (called 'ñire' in the Mapuche language). This association is found in east–west running valleys and near the upper forest limit. The araucaria canopy emerges from a prostrate and shrubby layer of ñire. Currently this forest type is an important source of fuelwood for Mapuche communities and is affected by grazing and by natural and/or man-made fires
- *Araucaria araucana* in pure stands is found near the upper forest limit on poorer soils, and also at lower altitudes interspersed in a steppe environment. Forest fragments on the steppe occur in isolated woodlots that are heavily disturbed by human activities (such as seed collecting, livestock grazing, etc).

2.2 Model design

The Pehuén Model was designed to answer a specific question: is it possible to simultaneously increase the regeneration capacity of the forest while also improving the livelihoods of the local people that depend upon it?

Model design followed four steps, as outlined by Randers (1980): conceptualization, formulation, testing and implementation. During Step 1, conceptualization, the purpose of the model is defined and its boundaries and key variables are identified. The behaviour of a modelled system is dependent upon its structure and the elements included. Thus, establishing the system's boundaries precedes the identification of key variables and processes that are also included in the model (Haraldsson 2000). The model should include all the elements that interact to make it dynamic (Cover 1996). If the causes of a system phenomenon or process lie outside the system as it is currently described in the model, then it is necessary to expand the original boundaries to encompass it. It is also necessary to describe comprehensible units of the key variables and to define the causal relationships within the system that are represented through feedback loops.

In Step 2, formulation, the diagrams that visually describe feedbacks are drawn and converted into level and rate equations. In Step 3, testing, parameter values are estimated and the model is tested using simulations based on the system's present conditions, but projected over a defined future period of time. Finally, in Step 4, the model is implemented to simulate system responses to management alternatives.

Feedbacks within the system are described using causal loop diagrams (CLDs). Each component of a system acts as a cause or an effect in related processes (Haraldsson 2000). CLDs are maps that represent this behaviour by illustrating the relationships between and among components of a system. A feedback represents a cascading process where an initial event ripples through a causation chain ultimately to affect itself (Martin 1997). There are two kinds of feedback: positive and negative. A positive feedback occurs when compounding, reinforcing or amplifying processes produce an exponential behaviour. A negative feedback drives balanced or stabilized systems to produce either asymptotic or oscillatory behaviour (Haraldsson 2000). Figure 1 presents an example of a causal loop diagram with positive and negative feedbacks.

System levels, flows and rates are determined so that computer simulations can be run. Levels are quantities that accumulate over time, flows are movements in and out of levels, and rates control the change in levels per unit of time. For example livestock population is a level, and recruitment and mortality of livestock are flows regulated by rates that modify the number of livestock in the system. The feedback mechanisms for the Pehuén Model, as exemplified in Figure 1, were imported into the STELLA (Structural Thinking, Experiential Learning Laboratory with Animation; Richmond 1994) modelling environment in order to run a computer simulation. In STELLA, levels and changes in levels are expressed through stock (rectangular boxes) and flow (arrows) symbols (Figure 2, overleaf). A rate of change might be thought of as a faucet that controls the flow of water into a bathtub (Roberts 2001).

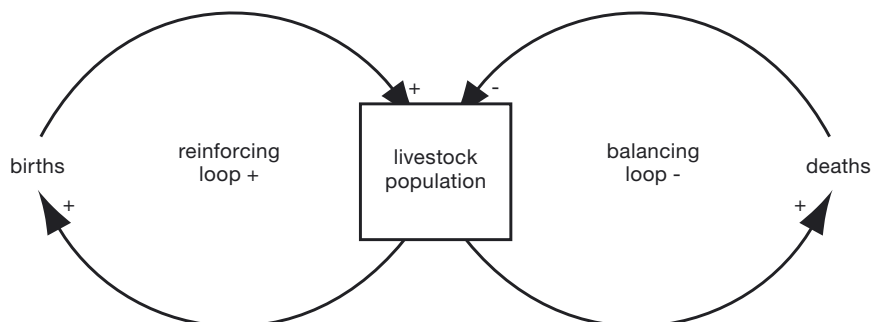


Figure 1. Example of a causal loop diagram (CLD) showing the positive (births) and negative (deaths) feedbacks to a livestock population.

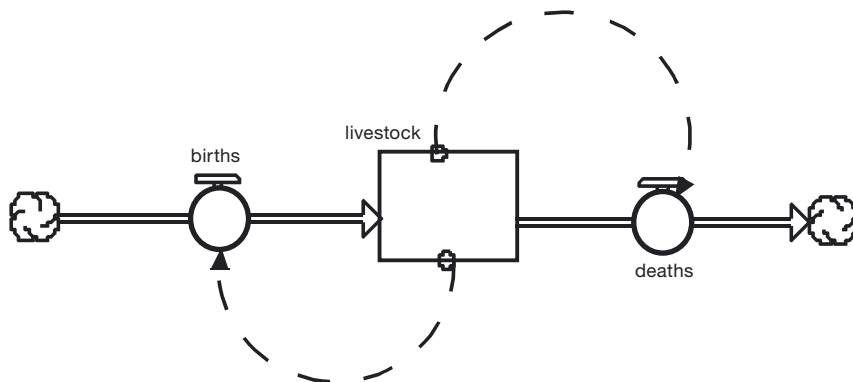


Figure 2. Relationships from the Figure 1 CLD displayed in STELLA stock and flow notations. Solid arrows (recruitment and mortality at their respective rates) flow in and out of the rectangular box, which represents the size of the livestock population. Dotted arrows indicate feedbacks from livestock population on flows.

2.3 Conceptualization of the model

The first step in designing a model is conceptualization (Albin 1997). It includes the following steps: (i) definition of the objectives of the model and its boundaries; (ii) identification of key variables; (iii) description of the behaviour of the variables; and (iv) description of the principal mechanisms or feedback loops of the system. The purpose of the Pehuén Model was to identify and analyze the relationships among the different elements, processes and players interacting within the araucaria forest ecosystem and, more specifically, to incorporate the impact of the indigenous communities on the forest and especially on the FGRs. Therefore, key ecological and socioeconomic variables were included in the system to help deepen our understanding of current araucaria FGR dynamics and to predict the evolution of the forest under future conditions.

Boundaries

The boundaries of the system corresponded to the boundaries of the Mapuche community of Chiuquilihuin (for more details on this study site, see Chapter 6). The area inhabited by the Chiuquilihuin Mapuche was identified as the most suitable for the study because its socioeconomic, ecological and forest genetic dynamics were representative of those found in other Mapuche communities. The size of the community territory is approximately 5000 hectares, located in the southern Andes in the Province of Neuquén in Argentina, between approximately 39°35'30"S and 39°39'30"S and 71°13'30"W and 71°0'0"W. To the north and west it borders Lanín National Park and to the south and east it borders other Mapuche communities and private ranches. The landscape is hilly and mountainous, ranging from 750 m asl to 2000 m asl. Precipitation decreases from 1800 mm to 1200 mm per year along a west–east gradient. *Nothofagus* spp. and *Araucaria araucana* are commonly found in association at higher elevations and in the more humid western areas they form dense forests. In the eastern part of the modelled area several fragmented but pure stands of *A. araucana* are found scattered in a steppe environment (800–1000 m asl) that is degraded by overgrazing (Figure 3). Little or no sexual regeneration was found within the araucaria woodlots in the steppe.

Subsystems

Three subsystems were identified as principally influencing the araucaria system within Chiuquilihuin territory: (i) community livelihood; (ii) seed availability for sexual regeneration of the forest; and (iii) livestock. The seed availability subsystem was divided into two



Figure 3. An isolated individual of *Araucaria araucana* in the steppe near the community of Chiuquilihuin (photo: A. Jarvis).

components as detailed data existed to describe them separately (see Chapter 6): seed production, which is based on the reproductive capacity of the mature trees, and seed consumption, which includes seed predation by animals and human collecting of seeds for a variety of purposes. The CLD diagram in Figure 4 describes the main variables and feedback mechanisms identified.

We characterized the genetics of the forest stands used for the modelling exercise and studied their gene flow. Gene flow appeared to move from the west towards the eastern fragments in the study area, with genetic information arriving via pollen from the continuous western forest stands of *A. araucana* and *Nothofagus* spp. However, there were indications that differentiation through genetic erosion would begin in the not-too-distant future in the eastern araucaria stands, which are pure but fragmented old-growth trees, now degraded by overgrazing. This will most likely result from a lack of natural sexual regeneration and will dampen the relatively high genetic diversity still found in the seed pools of these fragments. The likely effects of low or no sexual regeneration, restricted seed dispersal by rodents owing to disturbance caused by continued fuelwood exploitation, and the eventual removal of some mature individuals are described in the Figure 4 (overleaf) CLD in relation to the genetic processes of (assumed) endogamy and migration.

Research results generated for the seed availability subsystem allowed us to produce CLDs of seed production in relation to precipitation for the various araucaria forest stands included in the study. We found precipitation to be the principal factor controlling seed production in all the forest ecosystems investigated (see the simulation of seed production patterns over 50 years in Figure 9 p. 199). In fact, the number of seeds produced in any given year was determined by the annual average precipitation that occurred two years before. This could be related to the effect of precipitation during the pollination time (Sanguinetti *et al.* 2001).

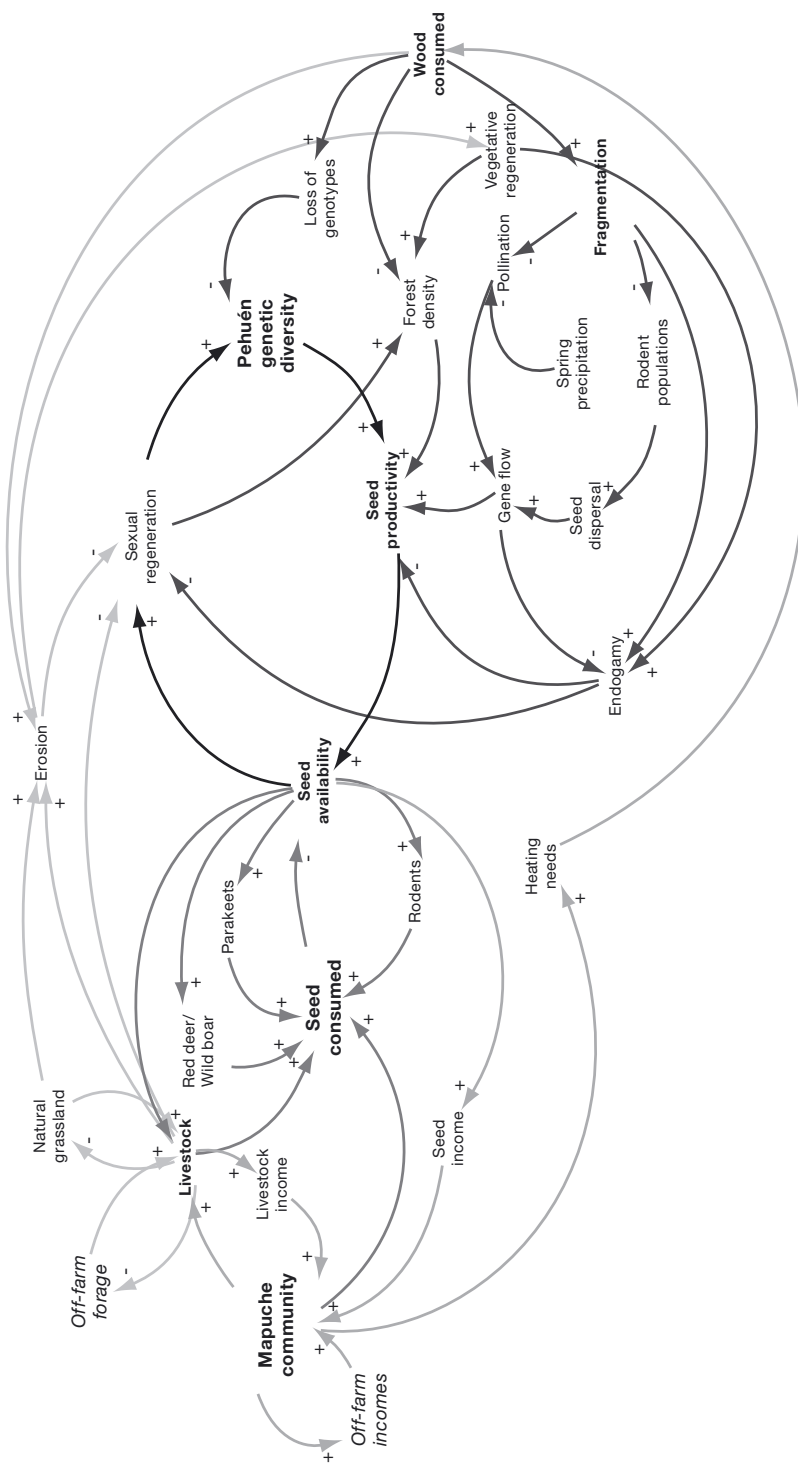


Figure 4. Causal loop diagram (CLD) showing the main feedbacks within the principal subsystems of the Pehuén Model.

In researching the seed consumption component of the seed availability subsystem, we found that seeds were being removed from the system by domesticated animals such as sheep, goats, cows and horses, by wild exotic animals like red deer and wild boar, by native rodents and parakeets, and by the collecting activities of the Chiuquilihuin Mapuche. Some seed predators were also involved in gene migration as they contribute to the dispersal of seeds, and this is also reflected in our conceptual model. Because araucaria seeds were used as livestock feed by the Chiuquilihuin community, livestock management decisions also affected seed availability and we attempted to incorporate these into the seed availability subsystem of our model.

The mechanisms by which the variables of the three main subsystems affect genetic diversity were identified through existing theoretical and experimental population genetic knowledge adapted to the characteristics of the pehuén system. For example, we assumed that araucaria forest fragmentation would have an indirect influence on genetic diversity by affecting gene migration. In the eastern araucaria forest, fragmentation increases the distance pollen must travel and/or decreases the available habitat for dispersers. At least at the time of our research, the genetic diversity of the fragmented araucaria populations was still comparable to that found in the denser forest stands to the west, with no evidence of increased endogamy and/or fixation or loss of alleles. Very likely, this was due to the sustained gene flow that was occurring along the west–east araucaria forest axis.

Soil erosion is another important system variable that has serious ecological, genetic and socioeconomic repercussions. Habitat loss resulting from soil erosion was occurring with varying degrees of intensity within the study area, and this was factored into our model. The most dramatic erosion occurs in the fragmented and isolated eastern araucaria stands that were surrounded by grasslands. The model reflects our conclusion that erosion affected regeneration and induced vegetative regeneration. We flagged this as a subject for future ecological research.

The community livelihood (or socioeconomic) subsystem was represented in our model with the following process components: livestock management, seed consumption, wood harvesting, income derived from seed, income from livestock, and income from off-farm activities. These components were analyzed and their variables quantified. In order to assess the sustainability of current livestock management practices, analyses of carrying capacity of the different grasslands scattered around the araucaria forests, and of their role in the pehuén system, were undertaken in collaboration with pastureland experts using geographic information system (GIS) tools from the Instituto Nacional de Tecnología Agropecuaria (INTA) in Bariloche, Argentina.

2.4 Indicators of sustainability

The amount of seed available for natural sexual regeneration of araucaria trees was identified as the key variable in the long-term stability of the pehuén system. Seed availability was strongly correlated with the maintenance of genetic diversity and, consequently, with the long-term evolutionary potential of araucaria forests. Minimum production of 18 000 seeds per hectare was determined to be the threshold for genetic sustainability. This value was obtained by combining estimates of the average germination capacity (70%) of seeds produced by several *A. araucana* populations in the study area, along with observation of the natural regeneration density of stands not affected by human seed collecting or by seed consumption by livestock or wild exotic animals. Under ideal conditions, a 70% annual araucaria seed germination rate would provide 12 600 seedlings/ha. This amount would be expected to secure araucaria regeneration and therefore *genetic* sustainability in araucaria forest evolutionary processes.

The indicator of the *ecological* sustainability of araucaria ecosystems was identified to be the forage productivity of wet meadows, one of several pastureland types found in the study area (see Table 6, Chapter 6). Wet meadows are the most productive, and are also the most resilient of the pastureland types to grazing pressure (see Section 3 below). Wet meadows were determined to be seriously degraded, and their carrying capacity consequently reduced, when forage productivity in relation to maximum productivity declined by 20% or more. This value represents the threshold for ecological sustainability.

Annual average per capita income of US\$500 (equivalent to 1500 Argentinean pesos/year) was identified as the principal indicator for sustainability of the *socioeconomic* subsystem. Any downwards change in average income is considered unsustainable.

3. Structure of the STELLA model

The three subsystems identified above were adapted into the STELLA modelling environment to allow a prediction of future conditions through simulation (Figure 5). Forage availability for livestock was modelled separately as data were available to represent the dynamics of pasturelands. The subsystems modelled are described below.

Livestock

This subsystem simulates the behaviour of livestock (sheep, goats, cows and horses; see Figure 6 (on p. 196) in the Pehuén Model. The modelling exercise could be refined by better describing pastureland degradation and regeneration processes, but this is a first attempt to describe livestock dynamics, and further research is needed to gather more detailed information. In order to standardize representation of grazing densities and the food requirements of the animals, a conversion into sheep livestock units (SLUs) was adopted (one SLU is a standard measure used to homogenize livestock numbers per surface area) in order to make comparisons possible. One SLU is equivalent to 1 sheep weighing 40 kg consuming 365 kg of dry forage per year (Siffredi *et al.* 2002). The conversion used was 1 sheep or goat = 1 SLU, 1 cow = 7.5 sheep or goats (or 7.5 SLUs), and 1 horse = 10 sheep or goats (10 SLUs).

The number of SLUs grazing in the study area was made dependent on annual variation in birth and mortality rates, and on the number of SLUs slaughtered annually. Livestock mortality was described in the model as partly controlled by the carrying capacity of the system, which corresponds to the ratio between the amount of food available on average every year and the amount of food consumed by one SLU. The amount of food available is calculated by converting the annual total production of all pasturelands into annual availability of dry forage, combined with pehuén seed availability. This latter value is calculated as the difference between seed production any given year and the amount of seed removed from the system by predators for consumption other than as livestock feed. Based on their nutritional values, Sanguinetti *et al.* (2001) converted araucaria seeds into equivalent dry forage units at a rate of 1 kg of seeds equalling 2.5 kg of dry forage.

The size of herds found in the pehuén system at any point in time depends on such variables as livestock management practices, the amount of meat consumed locally, trading opportunities for meat, the proportion of reproductive females in relation to the herds, and the amount of livestock food available, this last variable being the limiting factor. We observed that livestock in the pehuén system consumed per year on average 82% dry forage (from pasturelands) and 18% araucaria seeds (from araucaria forests). When dry forage was limited, the quantities of seed consumed increased, and this had negative effects on natural regeneration of araucaria forest stands and ultimately on the genetic diversity of those forests. This relationship demonstrates how araucaria genetic diversity is indirectly dependent upon the livestock management practices of the Mapuche community.

Forage availability

We determined the forage productivity in the five pastureland types and the number of seeds that could be sustainably harvested from araucaria forests (see Chapter 6). This enabled us to establish livestock carrying capacity for the pehuén system and to set a livestock density threshold above which overgrazing would occur. We also calculated a new variable that we called 'enhanced carrying capacity' that included livestock foods from sources external to the pehuén system, namely forage from summer fields in neighbouring areas outside our model boundaries, and hay supplied periodically.

The most productive pasturelands for forage are wet meadows. These are areas where water covers the soil or is present either at or near the surface of the soil all year round,

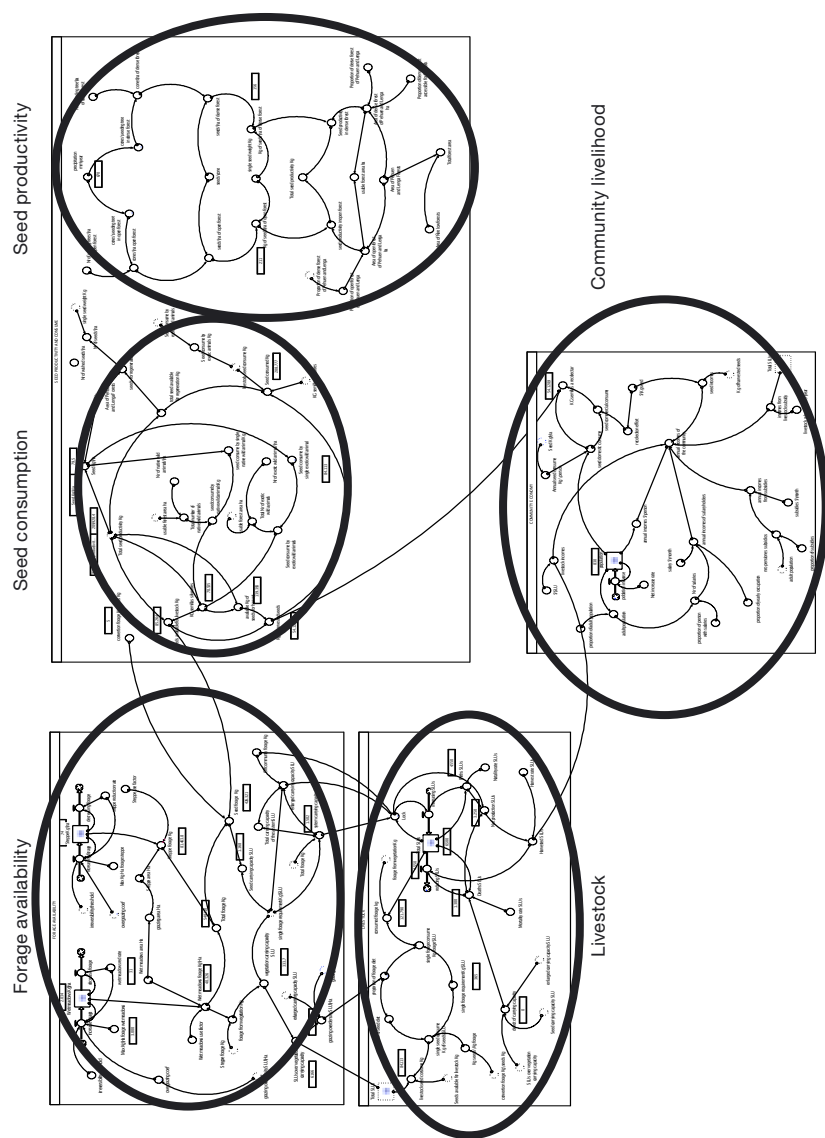


Figure 5. Overview of the STELLA stock and flow representation of the Pehuén Model, indicating the complexity of the system.



Figure 6. Mapuche people managing their livestock (photo: M.R. Lanari).

or for variable periods of time during the year. Maximum forage production grazed in the pehuén system was determined to range from 3000–5000 kg/ha of dry forage for wet meadows to 20–50 kg/ha of dry forage for the driest steppe pastureland type (Figure 7). A yearly proportion of production consumed by livestock (= use factor) was determined for these two pastureland extremes to be 65% and 40% respectively (see Annex 1; Siffredi *et al.* 2002).

The number of domesticated animals (= carrying capacity) that could be supported by the pehuén system was defined as the number of SLUs that could be fed sustainably by the combined supply of forage from the five pastureland types and from araucaria seeds. In our model, the availability of forage is linked to carrying capacity by a feedback mechanism. When overgrazing occurs, forage availability (pastureland forage and forest seed forage) declines along with carrying capacity. The equations that describe the relationship between the degree of overgrazing and the decline in forage productivity were formulated with the advice of experts from INTA.

Mapuche livelihood

Annual per capita income in Chiuquilihuin was chosen as the variable to be monitored to assess socioeconomic sustainability of the pehuén system. The size of the Chiuquilihuin community (306 people) and its net growth rate of 2% (including births, deaths, emigration and immigration; Pinna 2002) were used to initialize STELLA simulations (Annex 1).

Four sources of income were included in the model: livestock sales, araucaria seed sales, government subsidies and salaries. Subsidies and salaries were defined as off-farm sources. Annual per capita income from livestock was determined by calculating the volume of livestock sold by community members in a year (at current prices of 70 pesos or US\$25/SLU) divided by the number of community members. Annual per capita income from seeds was determined by calculating the amount of araucaria seed sold at 1.5 pesos or \$0.50/kg divided by the number of community members. The amount of seed collected any given year was set as a function both of seed availability and of motivation to collect, this latter variable assumed to be related to traditional domestic consumption



Figure 7. The Patagonian steppe ecoregion with its typical xerophytic vegetation, highly adapted to resist drought, wind and herbivores (photo: B. Vinceti).

rates and fluctuations in market prices. The amount of seed sold was calculated as a variable fraction of total seeds collected by the local community.

Off-farm income from subsidies was set as a function of the proportion of adult community members entitled to receive government subsidies: 30% of the population are adults, 75% of whom receive subsidies (Pinna 2002). Income from salaries was set as a function of the number of adults with temporary jobs (25% of adult community members, working for 7.2 months/year on average) and of the average monthly salary of 350 pesos or US\$120 (Pinna 2002). We determined average annual per capita income to be roughly \$500, around 1500 Argentinean pesos/year.

Seed production and consumption

A subsystem was designed to describe fluctuations in the number of seeds available for araucaria regeneration. This value was defined as the difference between the average annual number of seeds produced by the araucaria forest stands and the average annual amount of seeds consumed, which varied in relation to animal predation (Figure 8, overleaf) and quantities collected by the Chiuquilihuin Mapuche.

The average annual number of seeds produced was determined by the number of mature trees/ha and the annual precipitation in the preceding two years, reflecting the time it takes for the seeds to reach maturity. Rainfall amounts measured during four years of field research were used to extrapolate precipitation trends over a longer period. The number of cones/tree was inversely correlated with annual rainfall because we believe that high precipitation has a negative effect on pollination (Sanguinetti *et al.* 2002). Estimates of the number of seed trees/ha varied in our model in accordance with open versus dense pehúen forest settings. We also assumed that the negative effects of heavy rainfall on pollination would not be felt so much in open and drier forests as in denser, wetter forests (our model used 17 trees/ha to represent open, dry sites and 55 trees/ha for denser, wet sites; Sanguinetti *et al.* 2002).



Figure 8. Pre-dispersal seed predation of *Araucaria araucana* seeds by birds (photo: B. Vinceti).

The proportion of modelled forest area exposed to the actions of livestock, which include seed consumption, grazing and trampling of seedlings and resulting erosion, was estimated by field observations to be 30% of the total forest area. Different seed consumption rates were also attributed to red deer and wild boar ('wild exotic animals') found within Chiuquilihuin territory, and to native wild animals (rodents and parakeets). Seed consumers were assigned different behavioural and timing values. For instance, wild animals were assumed to be the first to feed on available seeds and then humans gathering seeds for food would further reduce seed availability, and finally livestock would consume the amount left, either by direct grazing or consuming seed fed to them by the Mapuche.

The amount of seed consumed by wild animals (both native and exotic) depended on their population sizes, on individual animal consumption rates, and on whether this was an area of forest where livestock grazing occurred. The amount of seed collected by local people or consumed by livestock and wild animals was also dependent on fluctuations in seed production.

Densities of wild animals were determined to be 50/ha for natives and 2/ha for exotics (Sanguinetti *et al.* 2001). Seed consumption rates by humans are described in the 'community livelihood' subsystem that focused on economic activities.

4. Pehuén Model behaviour under current conditions

4.1 Initial settings

Initialization values were attributed to 27 parameters incorporated into the Pehuén Model (Annex 1). Parameters are also defined as 'variable constants' or coefficients, that is they are quantities that are known and which the modeller may wish to change, but which must be given a value to start a modelling simulation. The equations that define the relationships between the 120+ variables (Figure 5, on p. 195) were formulated. The behaviour of the Pehuén Model was then tested under current management conditions. Simulations were run

over a 50-year period, which seemed a reasonable time for long-term forest management planning. This period also corresponded to the estimated time required for araucaria trees to reach reproductive maturity.

4.2 Results

The simulations revealed that current natural resource management practices in the pehuén system were not sustainable, whether viewed from genetic, ecological or socioeconomic points of view.

Annual system carrying capacity for livestock was found to vary strongly in relation to annual seed production, which in turn showed a significant negative correlation with the precipitation level two years prior to the year of simulation. Forage production from pasturelands within Chiuquihuin territory was less important than seed availability in contributing to livestock food needs.

Under current management practices, the simulation predicted that in 50 years, wet meadow forage productivity would steadily decrease to 60% of its initial value, showing that present conditions are ecologically unsustainable.

Under current conditions the simulation also showed that approximately 19 000 seedlings/ha would be found in dense araucaria forests unaffected by humans, while in open forests subject to anthropogenic pressures and grazing, there would be only about 500 seedlings/ha. In Figure 9, the cyclical peaks in araucaria seed availability, what we called ‘potential natural regeneration events’, are simulated over the 50-year projection period. Peaks in seed production occur at intervals of 3–5 years, reflecting low annual precipitation values two years before, but they never reach the minimum annual production of about 18 000 seeds needed to secure proper regeneration densities. We interpret this to mean that seed will not be available to ensure adequate natural araucaria regeneration in the pehuén system over the next 50 years if management practices remain unchanged.

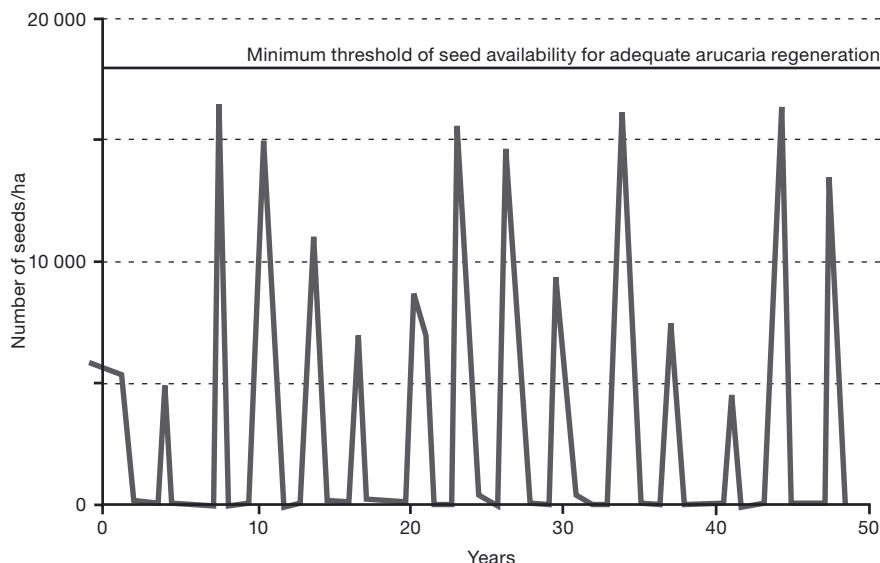


Figure 9. Number of seeds available for regeneration over the 50-year simulation. The upper horizontal line indicates the minimum threshold of seed availability sufficient to enable adequate regeneration events to occur. The simulated fluctuation in seed production is strongly correlated with variation in annual precipitation two years earlier.

A final simulation was run to estimate variation in annual per capita income over the 50-year future period. During this time, according to the current population growth rate, the Chiuquilihuin community is expected to increase to three times its current size (Figure 10). This growth should consequently increase araucaria seed consumption and, assuming a constant amount of SLUs in the community, reduce the annual per capita income by about 30% from US\$500 to \$330 (Figure 11). Over the same period, degradation of pastureland from overgrazing would lead to a reduction in available forage (mostly due to a reduction of the wet meadows forage production; Figure 12, on p. 202), negatively affecting livelihoods by increasing livestock mortality and decreasing income from livestock.

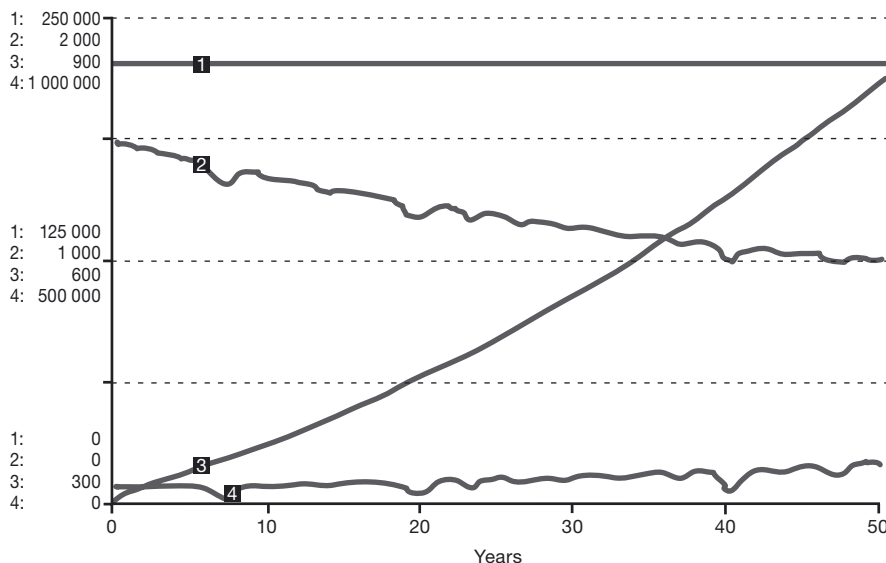
Thus, without some sort of significant change, such as government interventions leading towards sustainable management and/or adoption of conservation principles, the living conditions of local communities will deteriorate and the environment will become heavily degraded, with very serious loss of araucaria genetic diversity.

5. Sensitivity analysis

A sensitivity analysis was carried out to test how responsive the model was to changes in values of certain parameters. Sensitivity analyses help to identify variables that disproportionately affect the behaviour of systems and, for modelling purposes, the values of these variables should not be estimated but rather derived whenever possible from empirical observations (Breierova and Choudhari 1996). Nevertheless, in the present dynamic model, some variables were difficult to measure in the field and some relationships were difficult to predict, for instance, responses in araucaria seed collecting in relation



Figure 10. Mapuche children (photo: A. Pinna).



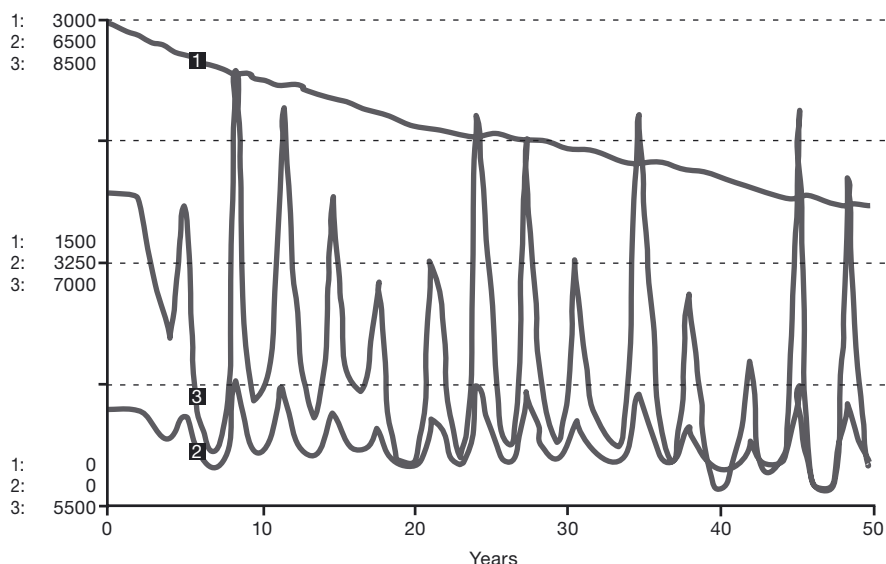
1. Annual income from livestock (US\$)
2. Annual income per person (US\$)
3. Increase in population (total number of inhabitants in the Chiuquilihuin community)
4. Annual income from sale of araucaria seeds (US\$)

Figure 11. Simulation of population increase over 50 years, based on the current population growth rate, and the consequent behaviour of annual per capita income. The number of sheep livestock units (SLUs) does not increase as it is constrained by forage availability (the maximum number of SLUs supported by the system has already been reached), therefore income associated with livestock will not increase over time either. Income from araucaria seed rises slightly owing to increased seed harvesting. Overall, annual per capita income (combining all sources) declines over the period of simulation.

to changes in seed price. Hence, the value of some parameters and some conversion coefficients were based on best estimates.

The sensitivity of the three primary indicators of genetic, ecological and socioeconomic sustainability (seeds available for regeneration, forage productivity of wet meadows, and annual per capita income) was tested. Analyses were also carried out to assess the sensitivity of the model to variations in forage productivity on the steppe, the fraction of dense forest accessible to livestock, the conversion coefficient of seeds into forage units, the total number of SLUs introduced into the modelled system, the birth and death rates of livestock, the proportion of dry forage in the livestock diet, the human population growth rate, the proportion of adults to total population, the percentage of adults that receive subsidies and salaries from government, and the amount of those salaries.

The default values for 15 of the parameters (shown in Annex 1) used for the simulation of Chiuquilihuin community dynamics were modified by $\pm 20\%$ to test the repercussions of these changes on the system, considering that a $\pm 20\%$ variation in the values was a realistic possibility that could occur for many reasons. The sensitivity analysis was undertaken without considering forage inputs from sources outside the system. Using the new values, for the 50 year projected future period, we recalculated the three primary indicators of genetic, ecological and socioeconomic sustainability and compared the quantities obtained to the corresponding results of the original simulations. The repercussions of these changes on the three selected indicators of sustainability were monitored, and



1. Annual forage production from wet meadows (kg/ha)
2. Annual contribution of seed to feed livestock (SLUs/ha)
3. Total number of SLUs supported by overall food availability within the system (forage and seeds)

Figure 12. Simulation of the progressive decline in forage availability from wet meadows owing to protracted overgrazing (i.e., maintaining current exploitation rates of pasturelands) over the 50-year simulation period. Wet meadows represent a very small fraction of the overall pastureland used by the Chiuquihuin community, but they are extremely productive and contribute to most of the forage availability (their forage productivity is 30 times higher than that of the steppe). The total number of sheep livestock units (SLUs) supported by the food available within the modelled system fluctuates in strong correlation with seed production pulses.

changes of more than 5% from the original value were considered significant. Results are presented in Table 1.

The analysis provided both predictable and unpredictable results. For example, the genetic sustainability (seed available for regeneration) of the pehuén system was strongly correlated with seed production and consumption, as highlighted previously in the model diagram (Figure 4, on p. 192).

5.1 Discussion

If livestock grazing could, in part, be shifted to denser araucaria forests further from the Chiuquihuin villages, the amount of seed available for regeneration in the forests near the villages would increase because grazing would be distributed more equitably throughout the pehuén system. Even though this change would result in a more extensive grazing regime, the fact that the new forests now being exploited have higher seed productivity and are further from the villages means that the forests closer to the villages would be subjected to lower grazing pressure, reduced seed consumption, and thus exhibit more regeneration.

However, the same response could result from any variation in the system that would reduce seed consumption, such as an increase in the proportion of grass in the livestock diet or an increase in the nutritional efficiency of seeds, or both. In contrast, a rise in seed collecting efforts driven, for example, by an increase in seed prices or an increase in the

Table 1. Results of the sensitivity analysis of 15 variables in the Pehuén Model. The initial values of the variables were changed by $\pm 20\%$, producing changes in values of the three indicators of sustainability. The initial values of the principal Pehuén Model variables are presented in Annex I.

Variables modified to test the sensitivity of the three indicators of sustainability	Change in default values	Change in values of the three indicators of sustainability, expressed as percentages of the default values		
		Genetic Seed (kg) available for regeneration	Ecological Productivity of wet meadows	Socio- economic Annual income per capita
Forage productivity of steppe	-20%	-0.4%	0.0%	0.2%
	+20%	0.3%	0.0%	-0.1%
Dense forest accessible to livestock	-20%	-84.1%	0.3%	-2.2%
	+20%	196.7%	2.7%	1.7%
Fraction of forage in livestock diet	-20%	-27.2%	0.0%	0.0%
	+20%	+38.9%	0.0%	0.0%
Total number of SLUs	-20%	24.0%	4.5%	0.4%
	+20%	-7.3%	-7.3%	-0.04%
Human population net increase rate	-20%	+10.2%	0.0%	0.5%
	+20%	-10.7%	0.0%	-0.5%
Livestock mortality rate	-20%	-9.2%	-0.6%	0.2%
	+20%	9.2%	0.3%	-0.2%
Conversion factor (seed to forage)	-20%	-9.3%	-0.3%	-0.1%
	+20%	+8.5%	0.0%	0.1%
Pehuén seed price	-20%	4.7%	0.0%	-1.0%
	+20%	-8.8%	0.0%	1.2%
Seed collection effort	-20%	5.5%	0.0%	-0.2%
	+20%	-5.4%	0.0%	0.2%
Monthly salary (pesos)	-20%	0.0%	0.0%	-5.6%
	+20%	0.0%	0.0%	+5.6%
Fraction of people with salaries	-20%	0.0%	0.0%	-5.6%
	+20%	0.0%	0.0%	+5.6%
Fraction of adults in local population	-20%	0.0%	0.0%	-18.4%
	+20%	0.0%	0.0%	+18.4%
Fraction of adult population receiving subsidies	-20%	0.0%	0.0%	-12.8%
	+20%	0.0%	0.0%	+12.8%
Livestock birth rate	-20%	0.0%	0.0%	-0.8%
	+20%	0.0%	0.0%	0.8%
SLU price	+20%	0.0%	0.0%	-0.6%
	+20%	0.0%	0.0%	0.6%

number of SLUs in the system, would have a strong negative effect on the availability of seed for natural regeneration (genetic sustainability indicator).

Interestingly, the sensitivity analysis showed that the system would be unaffected by an increase in the price of livestock. This might be explained, in part at least, by management constraints. The majority of the families in Chiuquilihuin own on average about 40 SLUs, and this number seems to represent an upper limit of animals that average families can manage for their own needs under their specific land ownership situations.

Finally, increasing the number of animals in the community negatively affected natural araucaria regeneration by reducing seed availability. It also led to a depression in forage productivity in wet meadows owing to overgrazing. In contrast, if just 20% of the current 6500 SLUs were removed, substantial benefits would be achieved in genetic sustainability.

6. Alternative management scenarios for the pehuén system

Based on the results of the sensitivity analysis and on the knowledge we obtained through field research, we simulated several hypothetical management alternatives and monitored their outcomes over a 50-year period in terms of genetic, ecological, and socioeconomic sustainability. The evaluated alternatives were:

- 1a. Adjust livestock numbers to pastureland carrying capacity, or increase forage inputs
- 1b. Reduce grazing pressure (total SLUs)
2. Increase araucaria seed prices
3. Increase off-farm incomes.

1a. Adjust livestock numbers to pastureland carrying capacity, or increase forage inputs

The ability of pasturelands to recover from grazing pressure was assessed by simulating regulatory regimes that maintained livestock numbers within the limits of pastureland carrying capacity. We began the simulation by using current rates of use and current conditions of pastureland degradation, but with no additional sources of off-farm forage. We found that over a simulated period of 8 years, the pastureland productivity declined to a point where the number of SLUs that could be sustained dropped from 6500 to approximately 940. This reduction in SLUs progressively relieved pastureland grazing pressure and, when maintained at this level, a slow but steady recovery in forage productivity occurred after another 8 years, especially in the wet meadows – our ecological sustainability indicator. Under these conditions, maximum pastureland productivity was restored near the end of the 50-year simulation period. The positive repercussions of reduced grazing pressure at this rate and magnitude could also be seen in more frequent regeneration pulses, because seed availability for natural regeneration now more often passed the minimum threshold for sustainability. While this scenario seems to achieve genetic and ecological sustainability in the pehuén system, it is economically unrealistic over the simulation period, both because of current population growth and because income reductions from lower livestock numbers would presumably be unacceptable to the Chiuquilihuin community.

Nevertheless, other options exist that could maintain the current number of SLUs without causing lowered pastureland productivity, such as: (i) extending grazing areas to additional pasturelands located outside the community land that are not currently being used, (ii) increasing hay supplied by the provincial government during difficult times, (iii) implementing more intensive pastureland management practices, particularly in the wet meadows, by employing fences to regulate grazing, or (iv) by trying to restructure use of the wet meadows that are owned by a few families, by somehow making them more accessible to other community members (Siffredi *et al.* 2002). For example, with all other variables remaining unchanged, the simulations showed that by doubling off-farm forage inputs to 3 million kg of dry forage per year, the productivity of wet meadows recovered over a span of 12 years. However, this scenario did not result in natural araucaria regeneration nor increases in per capita income over time. Moreover, some of the management alternatives

simulated have practical limitations in that they conflict with traditional practices, which is the case with using fences, as well as with current property ownership patterns. We concluded that improvements in pastureland management should not be considered in isolation but rather in conjunction with other solutions.

1b. Reduce grazing pressure (total SLUs)

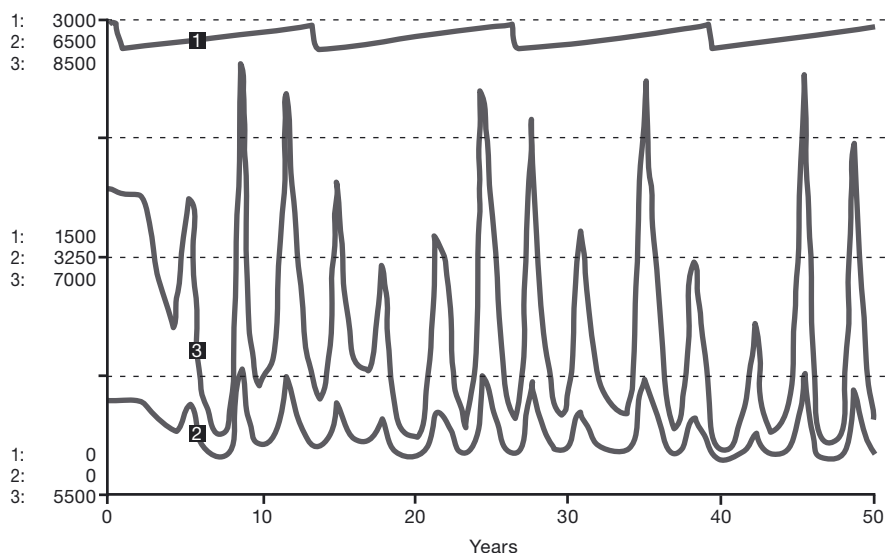
The Mapuche of northern Patagonia have practised livestock transhumance for more than 400 years, adapting themselves to a pastoral way of life (Lanari *et al.* 2003). During the BMZ-funded project, we found livestock to be unequally distributed among Mapuche families (Pinna 2002), but most families owned on average about 40 SLUs. We called this value 'cultural stock' because it corresponded to the optimal number of animals most families chose to raise. Only two farmers owned more than 200 animals, while nine farmers owned between 51 and 150 animals (Pinna 2002). In the scenario simulated, the average cultural stock figure was multiplied by the number of families in Chiuquihuín, thereby reducing the total community livestock holdings to 2520 SLUs, which represented about one-third of the 6500 SLUs now held by the community. This calculation reflects a hypothetical condition in which all farmers own an equal number of animals. When the number of SLUs is so reduced, grazing pressure diminishes and this triggers a fast recovery of wet meadow productivity. At the same time the frequency of *araucaria* regeneration events (seeds available above the regeneration threshold) increased. However, reducing the number of SLUs induced an immediate 30% decrease in annual per capita income and halved annual per capita income after 50 years. This alternative was thus not sustainable from a socioeconomic point of view.

2. Increase *araucaria* seed prices

The management options thus far proposed have shown that by changing pastureland management and reducing grazing pressure, ecological and genetic sustainability can be achieved. However socioeconomic sustainability could not be achieved because of income losses from fewer SLUs combined with projected population growth. Therefore, we simulated alternatives that increased income using other products and income sources. We ran a simulation that maintained SLUs at 2520 while doubling income from *araucaria* seeds and keeping off-farm incomes unchanged. The simulation showed that this change was initially sustainable in genetic, ecological and socioeconomic terms, but in the long run higher seed prices combined with local population growth led to rates of seed harvesting that rapidly depressed *araucaria* regeneration. After ten years and thereafter, seed available for natural regeneration never reached the minimum threshold for genetic sustainability. Moreover, the temporary increase in income generated by higher seed prices did not balance the income losses from the reduction in SLUs. Therefore, an increase in seed prices was seen as unlikely to produce sustainable outcomes in the long run.

3. Increase off-farm incomes

A simulation was run with community livestock holdings again set at 2520 SLUs but now with average annual off-farm income increased from 350 pesos (US\$120) to 500 pesos (\$160). And we assumed that each family would benefit from at least one salary all year round. Thus, our original 'fraction of inhabitants with salaries' per family (see Annex 1) grew from the original value of 0.25 to 0.68, and the 'period during the year with employment', expressed as a fraction of a year, increased from 0.6 to 1. These relatively small and reasonable changes generated an immediate increase of 30% in annual per capita income. Combined with genetic and ecological sustainability achieved through reduced grazing pressure (Figure 13, overleaf), this socioeconomic change could secure long-term stability in the pehuén system. But because of projected population growth, off-farm incomes would need to rise over time in order to maintain the long-term socioeconomic, genetic and ecological benefits.



- 1: Annual forage production from wet meadows (kg/ha)
- 2: Annual contribution of seed to livestock feed (SLUs/ha)
- 3: Total number of SLUs supported by overall food availability within the system (forage and seeds)

Figure 13. Simulation of forage productivity of wet meadows following an increase in off-farm incomes. Forage productivity of wet meadows displays small fluctuations and does not decline dramatically over time as projected under current livestock management conditions. The total number of sheep livestock units (SLUs) supported by overall food availability within the system is tightly linked to pulses of seed availability over time, but periodically reaches higher values than under current conditions (i.e., without an increase in off-farm incomes).

7. Conclusions

Linear thinking or strictly deterministic analytical tools do not permit us to understand the complexity of human-modified ecosystems like those found in the araucaria forests of northern Patagonia. For this study, a systems approach was very helpful in representing the dynamic interdependencies among the many parameters and variables that were needed to model the functioning of araucaria forests in the area investigated.

Our simulation revealed expected and unexpected dynamic behaviour, and it highlighted the need to undertake more detailed studies of links between genetic, ecological and socioeconomic aspects of the pehuén system. In future, we will particularly need to fill data gaps and to improve our understanding of several feedback mechanisms within the system. Nevertheless, using current data and trends to extrapolate conditions over the next 50 years, our simulation showed that the pehuén system would become less-and-less sustainable in forest genetic, ecological and socioeconomic terms. Indeed, over our 50-year simulation period, using current natural resource use patterns, further degradation of pasturelands and less and less forest regeneration will occur. This projected future scenario, when coupled with the projected growth in the Mapuche population, causes us to be seriously concerned about the future prospects of the pehuén system.

The sensitivity analysis and the simulated management alternatives led to the identification of four variables as most important in moving the pehuén system towards sustainability:

- Number of SLUs
- Seed production in araucaria forests

- Rate of population growth in the Chiuquilihuin community
- Off-farm incomes.

The factor exerting most pressure on the sexual regeneration of araucaria forests and, therefore, indirectly on araucaria forest genetic diversity, proved to be livestock grazing, and most of the scenarios we modelled showed that reducing the number of SLUs solved this problem. But while reduced grazing pressure lead to genetic and ecological sustainability, projected population growth and loss of income from fewer SLUs meant that socioeconomic sustainability was not achieved. Indeed, without some sort of significant compensation for income losses, reducing herd sizes would be unacceptable to the Chiuquilihuin community. Livestock is not only a key resource but is a traditional part of the Mapuche way of life.

Thus, our simulations showed that annual per capita income must be maintained and progressively increased at the same time that genetic and ecological sustainability solutions are being implemented. The most likely way to achieve this is through increasing off-farm incomes in the system.

However, even if more efficient livestock management could be achieved and complemented with alternative economic opportunities and without a reduction in the amount of SLUs, it would not guarantee genetic, ecological and economic sustainability over the long run. Such a situation indicates that political decisions are needed to achieve the two interlinked objectives of improving Mapuche socioeconomic conditions while conserving the evolutionary potential of pehuén forests. Measures implemented should be applied within a framework that respects local Mapuche culture and community traditions. We recommend that the government provide salaries for environmental monitoring and protection in compensation for the role historically and presently played by local families in conserving the araucaria ecosystems, as well as compensating local communities for the loss of income that would result from the recommended reduction in the number of SLUs.

At least one member from each Chiuquilihuin family could be officially recruited to work on community-based environmental protection and restoration projects. This would also contribute to raising awareness of conservation issues. Such government investments in environmental protection and indigenous community development might draw resources from wealthy economic sectors such as tourism, petrol and gas extraction, fruit orchards or mining. Investments in technology and institutional development should also be planned for this region.

References

- Albin S. 1997. Building a system dynamics model part 1: Conceptualization. Road Maps: A Guide to Learning System Dynamics. MIT System Dynamics in Education Project. Massachusetts Institute of Technology, Cambridge, USA.
- Armesto, J.J., C. Villagran and M.K. Arroyo. 1997. Ecología de los bosques nativos de Chile. Ed. Universitaria, Universidad Nacional de Chile, Santiago, Chile.
- Breirova L. and M. Choudhari. 1996. An introduction to sensitivity analysis. Road Maps: A Guide to Learning System Dynamics. MIT System Dynamics in Education Project. Massachusetts Institute of Technology, Cambridge, USA.
- Cover, M.J. 1996. Introduction to System Dynamics. Powersim Press, Resto, USA.
- Haraldsson, H.V. 2000. Introduction to system and causal loop diagrams. System Dynamic Course, Lumes, Lund University, Sweden.
- Lanari, M.R., H. Taddeo, E. Domingo, M. Pérez Centeno and L. Gallo. 2003. Phenotypic differentiation of exterior traits in local Criollo goat population in Patagonia (Argentina). Arch. Anim. Breed. 46(4):347-356.
- Martin, L. 1997. Introduction to Feedbacks. Road Maps: A Guide to Learning System Dynamics. MIT System Dynamics in Education Project. Massachusetts Institute of Technology, Cambridge, USA.
- Pinna, A. 2002. Informe de visita de campo y resultados de la encuesta demográfica y socioeconómica. Pp. 4. in Conservación, Manejo y Uso de Sustentable de los

- Recursos Genéticos de la *Araucaria araucana* en Argentina, Comunidades Aucapán y Chiuquilihuin, S.C de Bariloche, (L. Gallo, ed.). International Plant Genetic Resources Institute, Rome, Italy.
- Randers, J. 1980. Elements of the system dynamic method. Pp.117-139 *in* Building a System Dynamics Model Part 1: Conceptualization (S. Albin, ed.). Productivity Press, Portland, USA.
- Rechene, C. 2000. Los bosques de *Araucaria araucana* en Argentina. Estudios silvícolas. Centro de Investigación y Extensión Forestal Andino Patagónico, Esquel, Chubut, Argentina. Lehrstuhl für Waldbau und Forsteinrichtung, Universidad Técnica de Munich, Freising, Germany.
- Richmond, B. 1993. System thinking: critical thinking skills for the 1990s and beyond. *Syst. Dynam. Rev.* 9(2):113-133.
- Richmond, B. 1994. Systems thinking/system dynamics: Let's just get on with it. *Syst. Dynam. Rev.* 10:135-157.
- Roberts, N. 2001. Levels and rates. Introduction to computer simulation, a system dynamics modelling approach. *In*, Road Maps: A Guide to Learning System Dynamics. MIT System Dynamics in Education Project. Massachusetts Institute of Technology (MIT), Cambridge, USA.
- Sanguinetti, J., L. Maresca, M. Gonzalez Peñalba, L. Chauchard and L. Lozano. 2002. Producción bruta de semillas de *Araucaria araucana*. 3er Informe Interno. Administración de Parques Nacionales, Argentina.
- Sanguinetti, J., L. Maresca, L. Lozano, M. Gonzalez Peñalba and L. Chauchard. 2001. Producción bruta de piñones de *Araucaria* y estudio de la regeneración. Segundo Informe Interno. Administración de Parques Nacionales, Argentina.
- Siffredi, G., J. Ayesa, C. Lopez and F. Izquierdo. 2002. Aptitud de pastoreo de los pastizales en las comunidades de Aucapán y Chiuquilihuin (Neuquén). Grupo de Pastizales, Laboratorio de Teledetección-SIG y Unidad de Genética Forestal. Comunicación Técnica Nro 91. Área de Recursos Naturales EEA Bariloche. Pp. 11 *in* Conservación, Manejo y Uso de Sustentable de los Recursos Genéticos de la *Araucaria araucana* en Argentina, Comunidades Aucapán y Chiuquilihuin, San Carlos de Bariloche (L. Gallo, ed.). International Plant Genetic Resources Institute, Rome, Italy.
- Veblen, T.T., T. Kitzberger, R. Villalba and J. Donnegan. 1999. Fire history in northern Patagonia: The roles of humans and climatic variation. *Ecol. Monogr.* 69(1):47-67.
- Young, A., D. Boshier and T. Boyle (eds.). 2000. Forest Conservation Genetics: Principles and Practice. Commonwealth Scientific and Industrial Research Organization (CSIRO) Publishing, Melbourne, Australia and CAB International, Wallingford, UK.

Annex 1. Initial values of the Pehuén Model parameters

Livestock management		
Number of SLUs	(1)	6 500
Livestock natality rate	(1)	0.70
Livestock mortality rate	(1)	0.20
Number of SLU slaughtered annually	(1)	1.00
Forage availability		
Maximum dry forage production of wet meadows (kg/ha)	(2)	3 000
Maximum dry forage production of steppe (kg/ha)	(2)	90
Area of wet meadows (ha)	(2)	30
Area of steppe pastures (ha)		2 750
Dry forage from sources outside the community (kg)		1 450 000
Use coefficient of wet meadows	(2)	0.65
Use coefficient of steppe pastures	(2)	0.40
Chiuquihuín community data		
Number of inhabitants	(3)	306
Population growth rate	(3)	0.02
Fraction of adults in total population	(3)	0.30
Fraction of inhabitants with salaries, per family	(3)	0.25
Part of the year with employment (expressed as a decimal)		0.60
Monthly salary (pesos)		350
Fraction of adult inhabitants with subsidies	(3)	0.75
Monthly subsidies (pesos)		160
Annual subsidy for livestock raising (pesos/SLU)		1.00
Price of pehuén seeds (pesos/kg)		1.50
Price of livestock (pesos/SLU)		70
Seed production and consumption		
Area of <i>Araucaria araucana</i> – <i>Nothofagus pumilio</i> forest (ha)	(2)	2 642
Fraction (%) of dense <i>Araucaria araucana</i> – <i>Nothofagus pumilio</i> forest		0.80
Fraction (%) of <i>Araucaria araucana</i> – <i>Nothofagus pumilio</i> forest accessible to livestock		0.30
Exotic wild animals per hectare	(4)	2.00
Annual average precipitation (mm)	(5)	–

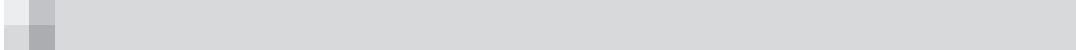
(1) Estimated from field observation

(2) Siffredi *et al.* 2002

(3) Pinna *et al.* 2002 and the pehuén project report

(4) Sanguinetti *et al.* 2001

(5) Not shown here. The 50-year series used in the model was extrapolated from precipitation trends starting from rainfall amounts measured during four years of field research.



Chapter 11

Environmental heterogeneity shapes genetic diversity through gene flow in *Araucaria araucana* forest ecosystems in Argentina

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1. Introduction

The results of spatial modelling are presented for one of the sites of the project funded by the German Federal Ministry for Economic Cooperation and Development (BMZ) on conservation, management and sustainable use of forest genetic resources. Modelling was carried out to determine how genetic diversity of the vulnerable pine tree *Araucaria araucana* is spatially distributed throughout the species' range in Argentina. This information is of primary interest to both conservation biologists (Escudero *et al.* 2003) and evolutionary biologists (Manel *et al.* 2003). The evolutionary forces of selection, gene flow, drift and mutation combine to shape the patterns of genetic diversity seen in nature. Modelling these forces can help explain and predict patterns of spatial genetic structure (Manel *et al.* 2003).

Although many models are available for analyzing genetic structure in one-dimensional selection clines (Barton 1999; Bekessy *et al.* 2002), they are of limited applicability to natural populations because most species inhabit selective environments that are complex and nonlinear. Here, we apply cline theory in a two-dimensional landscape analysis of selection pressures, gene flow and species distribution patterns to predict areas of high genetic diversity within the range of *A. araucana* (Molina). In eight populations of the species *A. araucana* we found a high correlation between observed levels of adaptive trait diversity and predictions based on the degree of environmental heterogeneity within the surrounding areas from which these populations can receive alleles through gene flow. This study provides a promising first step towards developing predictive tools for genetic conservation, both in *A. araucana* and other species, as well as a novel means of assessing spatial evolutionary processes.

2. The existing model

It is generally accepted that environmental heterogeneity acts as a diversifying force by providing many selection pressures to which a species must adapt (Hedrick *et al.* 1976; Linhart and Grant 1996; Nevo 2001). As such, one might expect that areas with the highest heterogeneity would tend to harbour the highest levels of genetic diversity within a species (Figure 1, overleaf, area A). While this relationship has often been demonstrated, it has typically been in primarily self-pollinating (Hamrick and Holden 1979) or asexual species (Bell 1997) that have low levels of gene flow.

This theoretical relationship is more complicated in species with high levels of gene flow because patterns of genetic structure can be spatially redistributed, resulting in increased diversity in relatively homogeneous areas (Slatkin 1973, 1978; Figure 1, right

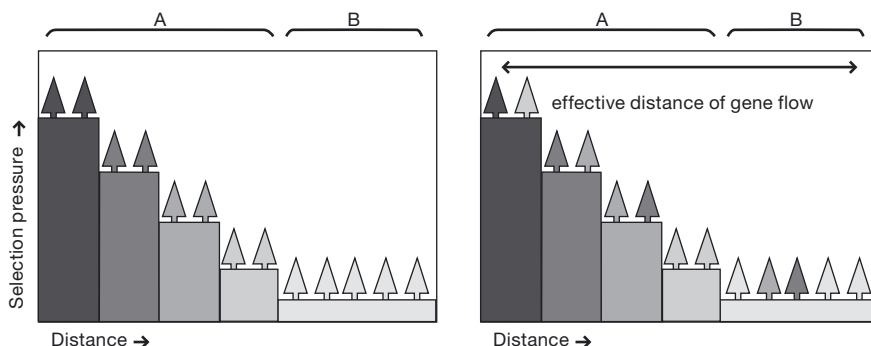


Figure 1. Theoretical genetic structuring in a variable landscape. Shaded bars represent the strength of selection pressure in each niche, while different shaded trees represent simple genetic adaptation to selection pressure. In the absence of gene flow (left), genetic structure is correlated to the local selection pressure, and diversity is high in the heterogeneous area (A), low in the homogeneous area (B) and low within niches. In the presence of gene flow (right), genetic structure is redistributed, resulting in increased levels of gene flow both within niches and in the homogeneous area (B).

panel, area B). Therefore, in studying how gene flow through a landscape shapes genetic structure, it is necessary to focus analysis on patterns within homogeneous niches.

Depending on the degree of environmental variability, a continuously distributed population may span several niches. In considering genetic structure within these niches, we focus analysis on subpopulations delineated by selection pressures rather than by demographic and geographic distribution. But because selection pressure is uniform within these homogeneous niches, any local diversity is the result of incoming gene flow from neighbouring regions that are differentially adapted (assuming no drift or mutation). In this context, one would expect the diversity of incoming alleles to be proportional to the variability in selection pressures affecting the donor subpopulations. Therefore, we hypothesize that the level of genetic diversity in a subpopulation inhabiting a homogeneous niche should be correlated to the heterogeneity in the environments of any genetically connected neighbours (we refer to this as ‘effective heterogeneity’ versus ‘simple heterogeneity’, which is measured within the same scale as diversity).

Because populations in nature are two-dimensionally distributed in space (unlike the simple model shown in Figure 1), we used maps of selection pressures and models of gene flow to represent two-dimensional clines (Figure 2).

3. Method

3.1 Measurements of diversity

Measurements of genetic diversity within populations were taken from a study by Bekessy and colleagues (2003), which assessed levels of variance in $\delta^{13}\text{C}$ and in the ratio of root mass to total plant mass. In this study, measurements were taken from seedlings grown in a common garden to control phenotypic plasticity. The dataset for diversity in RAPD (random amplified polymorphic DNA) markers comes from a different study by Bekessy and colleagues (2002), where leaf samples were taken for analysis from mature trees *in situ*.

3.2 Environmental variables

We prepared spatial grid maps representing the environmental variables using climatic datasets (Jones 1991) and geographic information systems (GIS – ESRI’s ArcInfo), with each cell representing an area of 100 x 100 m. We calculated three environmental variable

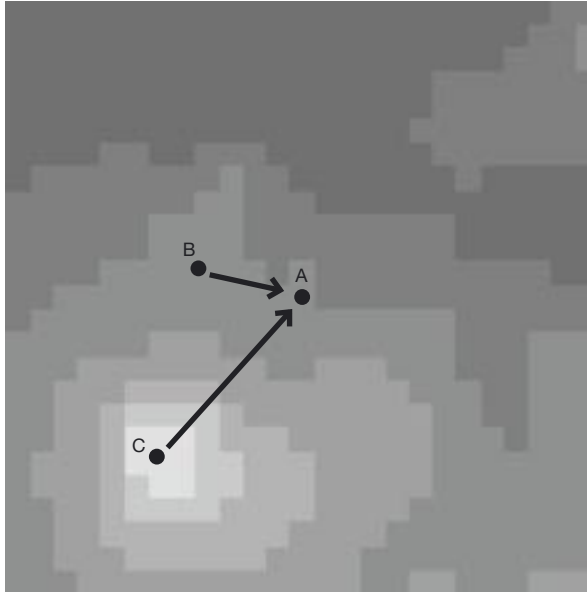


Figure 2. Two-dimensional selection-migration cline in a continuously distributed population. Here we measure the relative probability of gene flow from every individual cell of the subpopulation to the central cell (A), and calculate effective heterogeneity in (A) based on the diversity of environmental values in the region, weighted by the probability of genetic transfer from each. Assuming gene flow in the present example is distance-dependent, the environment in (B) would have a greater contribution to the calculation of heterogeneity than the environment in (C), as it is closer to (A). Each arrow is effectively approximating a one-dimensional cline.

maps: drought stress [DRT] from the number of quarter-month periods with less than 25 mm of precipitation, cold stress [CLD] from the minimum annual temperature, and ten randomly generated landscapes [RND] created using inverse distance weighted interpolation from 100 randomly generated points.

3.3 Measurement of heterogeneity

For each of the environmental variable maps we measured simple and effective heterogeneity using Simpson's index, a measure of diversity on a scale of 0, no diversity, to 1 (Simpson 1949). Effective heterogeneity is calculated for each cell in the map, and is a measure of regional variation in a given variable for the areas surrounding the cell. To measure effective heterogeneity, all cells within a 20.1 x 20.1 km area were included, the contribution of each surrounding cell being weighted by the probability of gene flow from that cell to the centre cell of the calculation. Since *A. araucana* has very low rates of seed movement (Bekessy *et al.* 2002), the probability distribution used to represent gene flow was based only on models of pollen transport (Tufto *et al.* 1997) combining distance effects using the one-tailed half-Cauchy distribution (Shaw 1995):

$$P(r) = (2 / \pi b) / [1 + (r / b)^2]$$

where b is the median distance of transport and r is the distance from the origin. Directional effects were represented by an average probability based on 'wind rose' data from three local weather stations (Fuerza Aerea Argentina 1975). Wind roses are polar coordinate datasets that show the frequency and wind speed for 360 degrees.

We set the median distance of pollen transport, b , at 200 m which has been found in *Quercus* spp. (Streiff *et al.* 1999) and should be generally representative of *A. araucana* because both are wind pollinated. We took population density into account by eliminating from the calculations all areas uninhabited by *A. araucana*, derived from a species distribution map that we created by reclassifying satellite images from paths 232 and 233, rows 86, 87 and 88 (Global Land Cover Facility 2003, <http://www.glcfapp.umd.edu:8080/glcfs/esdi>). These paths and rows correspond to the areas over which the Landsat satellite passes that are inhabited by *A. araucana*.

We calculated simple heterogeneity as explained above but used nonweighted probability distributions and measured heterogeneity within a radius of five cells around each central cell (since Bekessy and colleagues [2002] sampled 20 individuals at 100 m intervals, a radius of five cells [500 m] should be large enough to encompass all individuals in each subpopulation).

3.4 Statistical analysis

We calculated Pearson's r^2 correlations between subpopulation diversity and effective heterogeneity for each variable at each geographically corresponding cell. We calculated the P -value significance levels by bootstrapping r^2 correlations with 1000 repetitions (Bekessy *et al.* 2003). We calculated map correlations using the CORRELATION command in ArcInfo.

4. Results and discussion

To test the hypothesis described above, we compared existing datasets of variance in a physiological trait for drought avoidance ($\delta^{13}\text{C}$) in eight subpopulations of *A. araucana* (Bekessy *et al.* 2003) to relative measurements of heterogeneity in a spatially mapped dataset representing drought stress (DRT). Although it is difficult to evaluate whether the subpopulations sampled by Bekessy and colleagues occupied homogeneous niches, all individuals were sampled over small areas (see above) with very low levels of local environmental variation (maximum range of precipitation variability in the area occupied by any subpopulation was < 5% of the mean value for the area). We measured heterogeneity both within the area occupied by the subpopulations (simple heterogeneity, ~1 km²) and in the greater area from which they would be expected to receive pollen (effective heterogeneity, ~400 km²). We found high and statistically significant correlation between adaptive trait diversity and effective heterogeneity in drought stress ($r^2 = 0.828$, $P = 0.002$, $n = 8$; Figure 3) and found low, non-significant correlation with simple heterogeneity in drought stress ($r^2 = 0.030$; $P > 0.100$, $n = 8$).

Although generated from only eight samples, the high degree of significance in the former correlation suggests a strong relationship between fine-scale adaptive trait diversity and effective heterogeneity through selection pressure. Testing simple environmental heterogeneity allowed further confirmation that different selection pressures within the subpopulations are not shaping the levels of diversity, supporting the statement above that they are indeed occupying homogeneous niches.

We also tested the correlation between diversity of $\delta^{13}\text{C}$ and effective heterogeneity in both a random control variable (RND) and in a variable representing cold stress (CLD). Low, non-significant correlations with effective heterogeneity in RND ($r^2 = 0.259$, $P > 0.100$, $n = 8$) and in CLD ($r^2 = 0.463$, $P = 0.074$, $n = 8$) indicate that the correlations seen with effective heterogeneity in drought stress are effects stemming neither from the modelling process nor from nondrought-specific landscape heterogeneity. The higher correlation seen in CLD is likely the result of some similarity between the spatial distributions of cold and drought stress (we found medium map correlations between these variables, $r^2 = 0.317$). Similarly, fine-scale diversity seems to be density-dependent because correlations with heterogeneity calculated without modification by density (see above) were lower and nonsignificant in all variables ($r^2_{\text{DRT}} = 0.376$; $r^2_{\text{CLD}} = 0.001$; $r^2_{\text{RND}} = 0.017$; $n = 8$).

Taken together, these results show that the fine-scale levels of $\delta^{13}\text{C}$ diversity in *A. araucana* (Bekessy *et al.* 2003) are not a product of local environmental variation (because subpopulations inhabit locally homogeneous environments) but are dependent

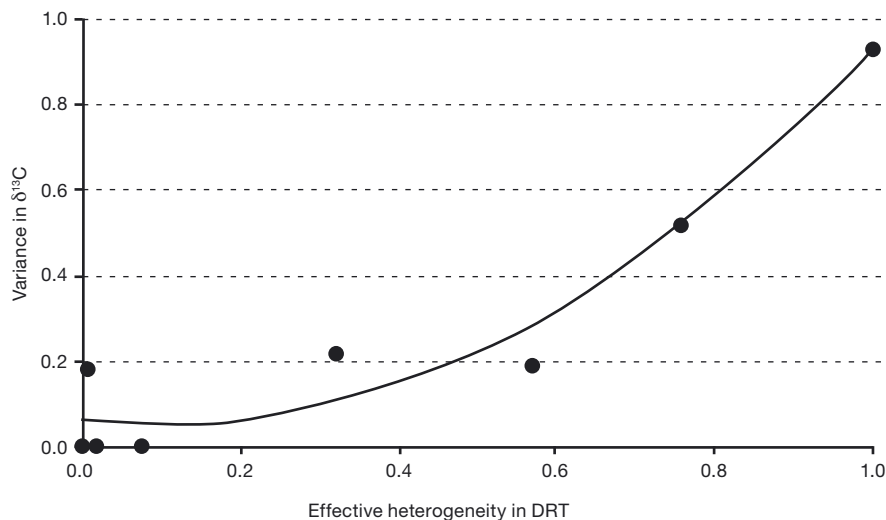


Figure 3. Relationship between effective heterogeneity in drought stress and variance in the adaptive trait for drought tolerance (DRT), $\delta^{13}\text{C}$.

instead on the inflow of genes from other differentially adapted subpopulations, supporting the model in Figure 1. Although gene flow in macro-evolutionary terms is typically seen as a homogenizing force opposing local adaptation (Lenormand 2002), these results support Slatkin's theory (Slatkin 1978) that it can also have locally diversifying effects through redistribution, depending on effective heterogeneity.

However, this relationship would not be expected to hold true for traits that are neutral or that experience only weak selection pressure, since factors such as genetic drift, mutation and 'hitchhiking' with other more heavily selected genes (Ohta and Gillespie 1996; Gillespie 2000) would more strongly affect their spatial distribution. Because heavy migration load (Lenormand 2002) can overwhelm local adaptation, and because the strength of this effect depends on selection pressure (May *et al.* 1975), weakly selected traits might not experience sufficient pressure to adapt locally. Patterns of genetic structure would not be expected to correspond with effective heterogeneity in the absence of local adaptation. Correlations between effective heterogeneity in drought stress and diversity in two other trait systems measured by Bekessy and colleagues (2002) were small and nonsignificant, probably because of their neutrality (in the case of RAPDs) or broad adaptive function (in the case of the ratio of root mass to overall mass [Bekessy *et al.* 2003]), which would be weakly selected.

While these tests of simple and effective heterogeneity show that pollen flow from surrounding regions drives fine-scale diversity, we cannot be sure that the model accurately represents the effective distances from which pollen actually is migrating. There was no way to test whether neighbours at different distances were actually contributing to diversity through gene flow. This effective range is probably dictated by factors such as the strength of directional selection purging nonlocally adapted alleles and the fragmentation of the species distribution, which could prevent long-distance multigenerational gene flow.

While the methods we used require further testing in other traits and species in order to increase confidence in them, the results have significant implications for both evolutionary and conservation biology. If these results describe a fundamental process applicable to all species, then effective heterogeneity in selection pressure should be well correlated to relative levels of genetic diversity in any strongly selected traits.

The methods described should therefore be applicable to predicting spatial patterns in the relative levels of diversity of traits that experience selection by stresses that can be spatially mapped. As such, the methods could help prioritize areas for collecting germplasm for *ex situ* conservation, while identification of genetically homogeneous areas could be important for ensuring that seeds for replanting are uniformly adapted to the areas from which they are collected. In addition, identifying effectively heterogeneous regions could help in designing conservation areas to protect genetic diversity. However, such recommendations would be of limited usefulness until the region – within which gene flow effectively contributes to local diversity – can be quantified. Because these methods use relative measures, the most diverse areas are not necessarily representative of the total range of variability in a given trait, so prioritization for conservation would also require accounting for absolute ranges of genetic diversity within the species. Besides, since patterns of heterogeneity will tend to differ from one selection pressure to another (as seen with DRT and CLD), patterns of diversity would require study on a trait-by-trait basis.

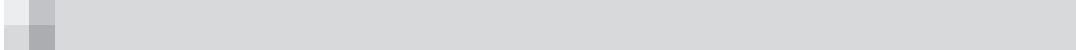
Many other analytical approaches developed to assess spatial patterns of genetic structure (e.g., Mantel tests and spatial autocorrelation, reviewed in Escudero *et al.* 2003 and Manel *et al.* 2003) are of limited applicability to studying adaptive traits because they focus on the effect of geographical distances rather than on selection pressures. While there has been some debate about the strength of correlation between adaptive and neutral diversity (Merilä and Crnokrak 2001; Reed and Frankham 2001; McKay and Latta 2002), evidence seems to favour a poor correlation. Since adaptive diversity is important to survival, methods applicable only to neutral markers may be of limited use to conservation planning. Additionally, where spatial methods have been applied to create map-based predictions (Hoffman *et al.* 2003), they have required existing genetic datasets to create predictions of diversity in new areas.

While the redistribution of alleles in migration-selection clines has been the subject of considerable investigation (Felsenstein 1976; Barton 1999), experimental studies have tended to focus on descriptions of the strength of selection, extent of gene flow, or shape of clines based on patterns in observed allele frequencies, and have been limited to examining these in species whose distribution spans approximately linear clines in terms of selection pressure (Mallet *et al.* 1990; Lenormand *et al.* 1999). We have approached the relationship from the opposite direction, predicting patterns of genetic diversity from spatial map-datasets of selection pressure and models of gene flow and then testing predictions against observed patterns of diversity. By using spatial map-datasets to represent highly complex landscapes of selection pressures, the methods used present a novel means of analyzing nonlinear two-dimensional clines. As such, they are applicable to the vast majority of natural populations that do not inhabit such simple environments. Moreover, since this method does not require delineation into what are sometimes arbitrary or biased *a priori* population designations (Manel *et al.* 2003), it is applicable to both continuous and fragmented species distributions. While it requires further testing and refining, spatial modelling of environmental heterogeneity and gene flow is an important approach both to understanding evolution in complex landscapes and to predicting patterns of adaptive trait diversity for conservation planning.

References

- Barton, N.H. 1999. Clines in polygenic traits. *Genet. Res. Camb.* 74:223-236.
- Bekessy, S.A., T.R. Allnutt, A.C. Premoli, A. Lara, R.A. Ennos, M.A. Burgman, M. Cortes and A.C. Newton. 2002. Genetic variation in the vulnerable and endemic Monkey Puzzle tree, detected using RAPDs. *Heredity*. 88:243-249.
- Bekessy, S.A., R.A. Ennos, M.A. Burgman, A.C. Newton and P.K. Ades. 2003. Neutral DNA markers fail to detect genetic divergence in an ecologically important trait. *Biol. Conserv.* 110:267-275.
- Bell, G.A.C. 1997. Experimental evolution in *Chlamydomonas*. I. Short-term selection in uniform and diverse environments. *Heredity* 78:490-497.

- Escudero, A., J.M. Iriondo and M.E. Torres. 2003. Spatial analysis of genetic diversity as a tool for plant conservation. *Biol. Conserv.* 113:351-365.
- Felsenstein, J. 1976. The theoretical population genetics of variable selection and migration. *Annu. Rev. Genet.* 10:253-280.
- Fuerza Aerea Argentina. 1975. Estadísticas Climatológicas: 1951-1960. Serie B, no 6. Fuerza Aerea Argentina. Buenos Aires, Argentina.
- Gillespie, J.H. 2000. The neutral theory in an infinite population. *Gene.* 261:11-18.
- Global Land Cover Facility. 2003. Earth Science Data Interface. University of Maryland, USA.
- Hamrick, J.L. and L.R. Holden. 1979. Influence of microhabitat heterogeneity on gene frequency distribution and gametic phase disequilibrium in *Avena barbata*. *Evolution* 33:521-533.
- Hedrick, P.W., M.E. Ginevan and E.P. Ewing. 1976. Genetic polymorphism in heterogeneous environments. *Annu. Rev. Ecol. Syst.* 7:1-32.
- Hoffmann, M.H., A.S. Glass, J. Tomiuk, H. Schmuths, R.M. Fritsch and K. Bachmann. 2003. Analysis of molecular data of *Arabidopsis thaliana* (L.) Heynh. (Brassicaceae) with Geographical Information Systems (GIS). *Mol. Ecol.* 12:1007-1019.
- Jones, P.G. 1991. The CIAT climate database version 3.7. Machine readable dataset of long-term climatic normals for the tropics. Centro Internacional de Agricultura Tropical (CIAT), Cali, Colombia.
- Lenormand, T. 2002. Gene flow and the limits to natural selection. *Trends Ecol. Evol.* 17:183-189.
- Lenormand, T., D. Bourget, T. Guillemaud and M. Raymond. 1999. Tracking the evolution of insecticide resistance in the mosquito *Culex pipiens*. *Nature* 400:861-864.
- Linhart, Y.B. and M.C. Grant. 1996. Evolutionary significance of local genetic differentiation in plants. *Annu. Rev. Ecol. Syst.* 27:237-277.
- Mallet, J., N. Barton, G. Lamas, J. Santisteban, M. Muedas and H. Eeley. 1990. Estimates of selection and gene flow from measures of cline width and linkage disequilibrium in *Heliconius* hybrid zones. *Genetics.* 124:921-936.
- Manel, S., M.K. Schwartz, G. Luikart and P. Taberlet. 2003. Landscape genetics: combining landscape ecology and population genetics. *Trends Ecol. Evol.* 18:189-197.
- May, R.M., J.A. Endler and R.E. McMurtrie. 1975. Gene frequency clines in the presence of selection opposed by gene flow. *Am. Nat.* 109:659-676.
- McKay, J.K. and R.G. Latta. 2002. Adaptive population divergence: markers, QTL and traits. *Trends Ecol. Evol.* 17:285-291.
- Merilä, J. and P. Crnokrak. 2001. Comparison of genetic differentiation at marker loci and quantitative traits. *J. Evol. Biol.* 14:892-903.
- Nevo, E. 2001. The evolution of genome-phenome diversity under environmental stress. *Proc. Nat. Acad. Sci. USA* 98:6233-6240.
- Ohta, T. and J.H. Gillespie. 1996. The development of neutral and nearly neutral theories. *Theor. Popul. Biol.* 49:128-142.
- Reed, D.H. and R. Frankham. 2001. How closely correlated are molecular and quantitative measures of genetic variation? A meta-analysis. *Evolution* 55:1095-1103.
- Shaw, M.W. 1995. Simulation of population expansion and spatial pattern when individual dispersal distributions do not decline exponentially with distance. *Proc. R. Soc. Lond. B. Biol. Sci.* 259:243-248.
- Simpson, E. H. 1949. Measurement of diversity. *Nature* 163:688.
- Slatkin, M. 1973. Gene flow and selection in a cline. *Genetics* 75:733-756.
- Slatkin, M. 1978. Spatial patterns in the distributions of polygenic characters. *J. Theor. Biol.* 70:213-228.
- Streiff, R., A. Ducousso, C. Lexer, H. Steinkellner, J. Glössl and A. Kremer. 1999. Pollen dispersal inferred from paternity analysis in a mixed oak stand of *Quercus robur* L. and *Q. petraea* (Matt.) Liebl. *Mol. Ecol.* 8:831-841.
- Tufto, J., S. Engen and K. Hindar. 1997. Stochastic dispersal processes in plant populations. *Theor. Popul. Biol.* 52:16-26.



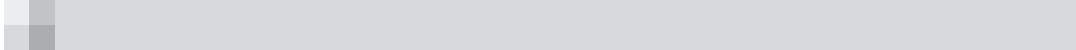


Part 3

Lessons Learned and Applicability of Research Outcomes

**The participatory approach in FGR
research within the BMZ-funded
project**

**The practical implications of
research outputs from forest
genetic studies**



Chapter 12

The participatory approach in FGR research within the BMZ-funded project

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1. Introduction

Classical approaches to conservation have often viewed people as threats to the preservation of natural ecosystems. Even as late as the 1970s, when many protected areas were being established around the world, management plans for these areas either ignored the local communities that depended on forest products for their livelihoods or were designed to exclude local people and what was seen as their harmful activities. As a consequence, many areas that were at the time accepted as being in the long-term conservation interests of the world community are now recognized as having come into existence at the expense of social and economic dislocations of local communities (Barrance 1996; Guha 1997). Only fairly recently has it become understood that protected areas can be both sustainably exploited and conserved, and in so doing avoid this conflict of interest. In fact, forest resources that include both timber and nonwood forest products (NWFPs) can actually contribute to conservation by providing local people with incentives to protect forests, while furnishing them with economically attractive alternatives to deforestation. Furthermore, such alternatives can help strengthen community resistance to external pressures to log forests or to convert forested land to other uses.

This pattern has several striking similarities with the way that plant genetic resources (PGRs) have been conserved and managed. Up until the early 1990s, the conservation of PGRs focused on *ex situ* approaches that mostly excluded the indigenous or peasant communities that had selected, preserved and used these same PGRs for generations. Since then, the involvement of local communities has been accepted more and more as an important variable in conservation and development success, and participatory approaches (PAs) are now widely applied, not only in PGR conservation, but also in programmes that emphasize PGR management and use (Almekinders and de Boef 2000). Indeed, strengthening community-level management of PGRs is now believed to be essential in order to make *in situ* conservation of crop genetic resources work and to guarantee that the community (members) share(s) in the benefits derived from the use of these resources. Farming communities also play important roles in improving PGRs for food and agriculture (FAO 1996) and PAs involving partnerships between conservation professionals and rural people in plant breeding and in the development of local seed supplies are now common elements of *in situ* PGR programmes worldwide (Ghimire and Pimbert 1997; Almekinders and de Boef 2000; Friis-Hansen and Sthapit 2000). A growing body of literature in rural development and natural resource conservation testifies to this evolution (Buchy *et al.* 2000).

The project in which we participated, funded by the German Federal Ministry for Economic Cooperation and Development (BMZ) and described in this volume, adopted

a participatory approach in studying forest genetic resource (FGR) conservation and management across a range of social and environmental contexts in Brazil and Argentina. We describe the framework adopted to assess the performance of the participatory methods and we then use this as a point of reference to analyze the effectiveness of the approach as it was applied at the four project sites. This perspective helped us to evaluate the degree to which partnerships were developed between researchers and local people in forest resource management and policy development. We conclude the chapter with a discussion of lessons learned, emphasizing both the need for capacity building among researchers and the need for longer time frames to implement participatory research and to develop effective partnerships.

2. A framework for participatory research

2.1 Concepts and definitions

Many PAs have been developed over the past 15 years with the goal of involving local people in research on agriculture, resource management and rural livelihoods. It is not possible to formulate a single easy definition for participatory research, as it needs to be designed and assessed within the context in which it occurs. It is better to understand participatory research as a collection of approaches that enables participants to develop their own understanding and control over processes and events being investigated (Ashby 2003).

The application of PAs to PGR research and agricultural development, including conservation, is thus context- and time-specific. Adaptation of a general participatory framework to different times and places requires flexibility and, as a result, variants in applied approaches have been developed (Halwart and Haylor 2001). Waisbord (2001) describes a fundamental conceptual difference between PAs: one type of approach only involves local communities in research and development 'as a means to an end', while the other views local involvement 'as an end in itself'.

In the first approach, participation of local communities is seen as a way of enhancing project effectiveness and impact; it corresponds with what Oltheten (1995) calls a target-oriented approach. Project objectives are defined in terms of outputs to be delivered by specific groups of stakeholders to 'outsider professionals', and participation is expressed in terms of the willingness of people to undertake and contribute to jointly identified activities.

In the second approach, also called a process-oriented approach (Oltheten 1995), project objectives are achieved when local communities choose to participate in the development process (Buchy *et al.* 2000). Participation is seen as an 'end' in a process where empowerment and (political) liberation of stakeholders occur (IPAP 1997). As such, it explicitly recognizes the role- and power-reversal processes associated with participation. In this second approach, participation means that the people themselves assume ownership and accountability for project activities, which they have identified and developed themselves with the support of 'outsider professionals' (Oltheten 1995), and which involves learned local responses to changes in society, economy and ecology (Röling and Wagemakers 1998).

The two approaches require different designs and implementation strategies. Each follows a separate path in determining the roles of participants who are most often research or other types of professionals and rural people (Buchy *et al.* 2000). Problems can arise when different participants have different expectations about the nature of the process and about project objectives. Yet, the two perspectives are not incompatible, and in practice their conceptual differences are not so clear-cut. Participation can therefore be seen as occurring on a continuum, where community involvement and mobilization might begin as a means to an end that eventually transforms itself over time into community empowerment, structural change and learned independent action. Similarly, projects might follow an approach that falls somewhere between a target- and a process-oriented approach. As we will see later in this chapter, the BMZ-funded project started as a more

target-oriented research project ('means to an end') but increasingly became process-oriented at several of the study sites ('end in itself').

2.2 Pros and cons of participatory approaches

The literature is rich in examples that describe positive and negative outcomes that have resulted from adopting PAs in development projects. One obvious positive feature of PAs is that they usually help improve researchers' understanding of conditions, problems and issues that are faced by target communities. As such, PAs can help ensure that researchers address the actual problems and needs of project beneficiaries. Directly engaging target communities and developing politically desirable objectives with them give legitimacy to projects and further the acceptance of their results and products within both local communities and wider audiences (IPAP 1997; Shannon 2002).

Based on lessons learned from case studies conducted in Africa, Asia and South America on participatory planning in community forestry projects, Oltheten (1995) concluded that PAs can also facilitate local empowerment by creating opportunities for especially disadvantaged groups to gain access to external resources such as training or credit, or to mobilize their own resources in the form of local organization, knowledge and skills. And positive outcomes like these enhance the capacity of local people to take action in defence of their own interests. Another advantage described by Oltheten is that PAs can lead to integration of local knowledge into project planning and implementation, thereby contributing to a greater likelihood of project success. Indeed, it would be fair to say that participatory planning results in a two-way learning and feedback process between local communities and researchers that improves the chances for timely adjustments of project support services to changing local realities. Moreover, the participation of local communities in research and development programmes strengthens capacity to identify and mobilize both local and external resources to undertake sustained action for positive change. Successful participatory planning can enhance political commitment and institutional support at many levels for local planning by building a common understanding between local groups and regional, national and international institutions.

On the other hand, projects that adopt PAs usually require more time, effort and resources in project development, trust-building and decision-making than do typical top-down development programmes. Moreover, in these projects, outputs may be less predictable, time lines for completing projects may need revision, and even the scope of projects can require modification. Changes like these can sometimes be difficult to reconcile within time-bounded or rigidly defined programme structures and funding mechanisms (Martin and Sherington 1997). Thus, participation may be a good long-term strategy but can have drawbacks when applied within short-term contexts or when issues need addressing urgently (Waisbord 2001). Nevertheless, the benefits of PAs can be great and they will usually outweigh disadvantages (Section 2.3 in this chapter). At the very least, PAs can substantially help local communities understand more about national and international decision-making processes that have the potential to affect project regions and beneficiaries, and this alone will usually have a long-lasting and positive effect on everyone involved.

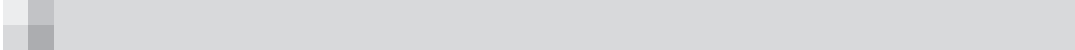
Lessons learned from the BMZ-funded project: objectives in using a participatory approach

In adopting a PA, the goal of the BMZ-funded project was to contribute simultaneously to sustainable development and to forest conservation through a deeper understanding of the effects of human activities on FGRs, in particular on the genetic diversity of selected target tree species. The outputs of the project's PA were information about and awareness of forest conservation, ecology and genetics that could provide local stakeholders with alternatives for managing their forests in a sustainable manner, and that could guide policy decisions in national and regional initiatives. A list of project objectives and possible impacts is presented in Table 1 (overleaf). The project goal was to achieve conservation

Table 1. Project objectives and intended impacts.

Disciplinary group	Project objectives
Socioeconomic research	<ul style="list-style-type: none">• Quantify the contributions of timber and NWFPs to household incomes and in relation to other income sources• Describe and quantify the effects of land-use (especially livestock grazing, collecting timber and NWFPs) on forest ecosystems and the genetic resources of key tree species• Examine changes in livelihoods resulting from social and economic change, including the consequences of continued use of FGRs• Assess the impact of current land use and forestry policies on the use of timber and NWFPs, and propose options favouring sustainable use
Genetic research	<ul style="list-style-type: none">• Examine the effects of human activities such as plant harvesting, logging and other anthropogenic pressures on the genetic diversity and the genetic structure of selected forest species, including NWFPs. This requires an understanding of patterns of genetic variation in selected species
Ecological research	<ul style="list-style-type: none">• Investigate the effects of human activities on reproductive biology and ecology, including phenology, pollinator foraging behaviour, seed dispersal, seed bank development, seed and seedling predation, and other regeneration dynamics in selected tree species
Development applications	<ul style="list-style-type: none">• Develop options for sustainable local management and use of selected trees and NWFPs under conditions of social and economic change, and position these options within a rational decision-making framework for managers and policy makers when they are considering changes to land-use policies and practices
Training, education and information dissemination	<ul style="list-style-type: none">• Strengthen national level planning and implementation of forestry genetic conservation programmes by providing training opportunities for scientists and representatives of stakeholder groups from the participating countries, and further training, education and information in project disciplines

Project impacts	Beneficiaries
Further development of an interdisciplinary base in conservation theory, methods and applications will be available to other countries wishing to initiate and/or to monitor <i>in situ</i> conservation of agricultural and forest biodiversity	National programmes (NPs) for agricultural biodiversity, any country
A better understanding of processes that affect genetic diversity will be available to guide national conservation and development programmes	National FGR programmes, any country
Examples, methodologies and guidelines will be provided that can guide other NPs in establishing <i>in situ</i> conservation projects	NPs for <i>in situ</i> conservation of FGRs, any country
Capacity of national programmes (NPs) to undertake conservation will be strengthened by the addition of scientific staff and by building formal links to community-based conservation efforts	NPs for FGR conservation in target countries
NP capacity for supporting <i>in situ</i> conservation will be strengthened in the fields of population biology, conservation biology, social science and ethnobotany. Capacity will be increased to analyze forest biodiversity using genetic markers, morphological assessments, social surveys and forest ecosystems research	NPs for <i>in situ</i> FGR conservation in target countries, especially science programmes
NPs will be strengthened in the sustainable use of local forest genetic resources. NPs in tree improvement and tree breeding will be improved	NPs for FGRs in target countries
Local communities and forest dwellers will incorporate conservation into their routine forest management activities, with connections to appropriate NPs	Selected local communities and conservation nongovernmental organizations (NGOs) in target countries
Local communities and forest dwellers will increase their capacity to use existing forest biodiversity in local development strategies through locally based forest management activities	Selected local communities in target countries
A better understanding of forest dwellers' biological knowledge and value systems, and the relationships of these systems with local patterns of FGR use and management	NPs for FGR use and conservation; international scientific community
Increased dialogue and partnerships among local communities, local governmental organizations, locally based NGOs and national agricultural research systems (NARS) to develop programmes for management	Local communities, NARS, local NGOs and governmental organizations



and sustainable use of FGRs and to assist local stakeholders in contributing to this goal. By achieving the project objectives, and in particular the fifth objective – training, education and information dissemination – we believed that stakeholders would be better equipped to make well-informed choices, and thereby become empowered to assume full control over the future management and use of their FGRs.

The project initially intended to follow a largely target-oriented approach by gathering data that would enhance our understanding of local resource use and management. This work was to be undertaken by the socioeconomic research component of the project. Involvement of local stakeholders in the genetic and ecological research activities was to remain limited. However, consultative meetings at the initial phase of the project led to substantially increased roles for various project stakeholders in the form of a continuous dialogue between researchers and these stakeholders. These meetings resulted in a gradual shift in project priorities, including the incorporation of activities tailored more to the needs of the local stakeholders. As we will discuss later in more detail, this enhanced consultation led to a wider acceptance of the project and more involvement of some stakeholders. Thus, from nearly the start of the project we began to experience a shift from a more target-oriented research project towards a process-oriented research approach.

2.3 Participatory approach to research

In the fields of applied agricultural science and agro-technology development, and especially in development-oriented research, the participation of rural people is essential for achieving sustainability (Reijntjes *et al.* 1992; Pretty 1995). Farmers can help scientists adapt agro-technology to specific environmental and agricultural settings, and they can help scientists apply external inputs more efficiently by drawing scientists' attention to their day-to-day decisions and actions with regard to those inputs (van Veldhuijzen *et al.* 1997). At the same time, scientists can shape research activities in order to meet farmers' needs and to elaborate best practices that incorporate locally generated innovation (Reij and Waters-Bayer 2001).

Three types of agro-technology development have been described in the literature (van Veldhuijzen *et al.* 1997). The first type occurs in what Braidwood (1967) has called an 'atmosphere of experimentation'. Rural farmers have practised this form of agricultural innovation from Neolithic times to the present, and today it is still practised in those parts of the world that are not greatly influenced by national agricultural development programmes. This type of agro-technology development is also called 'indigenous technology development'.

In modern times, and beginning especially in the late 19th century, national governments created agencies that specialized in agricultural research and disseminated their results through formal outreach efforts. These agencies operated according to a model in which agricultural technologies were developed in government research institutions and then transferred to farmers through extension programmes. This second type of agro-technology development is often called 'transfer-of-technology' (ToT), and for the last century it has dominated agro-technology development in the agriculturally advanced countries and in their policies of providing agricultural assistance to lesser-developed countries.

However, ToT programmes have not achieved the successes hoped for in developing countries, and in response to continuing criticism of the ToT approach, participatory approaches to agro-technology development were devised (Merrill-Sands 1986). This third type of agro-technology development, now called 'participatory technology development', stresses the combined importance of traditional farmers and scientists in agricultural innovation and change. When traditional farmers and scientists become partners in order to achieve common objectives, participatory technology development is viewed as a complementary approach to formal research and extension. Table 2 summarizes the main characteristics of the three types of agro-technology development.

Notwithstanding their advantages, PAs in agro-technology research are not always easy to implement because of what can be myriad inputs and demands from a variety of stakeholders. While the advantages of PAs are easy to understand in socioeconomic research projects, some biological scientists may be reluctant to adopt a PA in, for example,

Table 2. Characteristics of the three types of agro-technology development. Source: van Veldhuijzen *et al.* 1997.

Type	Indigenous technology development	Transfer of technology (ToT)	Participatory technology development
Objectives	Secure living, reduce risks	Maximize yield	Agricultural self-management by farmers, sustainability
Source of innovation	Farmers	Research organizations	Farmers partnering with research organizations
Nature of knowledge	Holistic	Particularistic	Creative tension between holistic and particularistic
Experimental approach	Largely unknown	Scientific procedures	Farmers' methods complemented by scientific procedures
Channels of information	Farmer-to-farmer	Extension services	Multiple systems: farmers, NGOs, extension agents, etc.
Process of communication	Informal, horizontal	Formal, vertical, top-down	Semi-formal, mixed direction
Role of rural people	Generators of knowledge, communicators, users	Receiver, adopter	Generators, communicators and evaluators of external ideas, users
Role of professionals	None	Teachers, controlling compliance with regulations	Multiple: facilitators, resource-people, co-researchers, trainers

a baseline study of population genetic dynamics, because the 'direct' involvement of rural people in such a study may be difficult for them to imagine (Section 2.4 this chapter). It is undeniable that some forms of basic research are better carried out in nonparticipatory ways unless great care is used in translating project methods and goals into terms accessible to all parties (Halwart and Haylor 2001). Nevertheless, in research projects that cover different fields, such as the ones described in this book, PAs can be more effective if applied by interdisciplinary teams of researchers and not individually by single scientists in their specific research area.

Moreover, even when local communities may not be able to participate in the implementation phase of all research activities, their input in the planning and prioritization of research activities may improve the relevance of the research for those communities and it may help both researchers and communities translate results into practice. The positive engagement of the public will almost always improve the chances for acceptance and adoption of recommendations derived from research activities.

Different phases can be identified in the unfolding of an 'ideal' PA in a research project: 1) participatory situational analysis – understanding problems and opportunities; 2) stakeholder identification; 3) consultation and negotiation on problem definition and project collaboration; 4) establishment of agreed-upon and transparent research strategies, objectives, responsibilities, roles and partnerships; 5) experimentation – data collection; 6) experimentation – data analysis; 7) sharing results and translation of results into appropriate outputs; 8) dissemination of results; and 9) sustaining the process (Jiggins and de Zeeuw 1992; van Veldhuijzen *et al.* 1997). Monitoring and evaluation mechanisms should be in place at all stages of research and involve relevant stakeholders.

Figure 1 illustrates the phases of an ideal participatory research and development project. Displayed are possible reiterative routes in initial phases, and then as each subsequent phase comes on line, more or other stakeholders participate and/or further agreements are reached.

Lessons learned from the BMZ-funded project: participatory approaches in the different phases of a research project

The participation of stakeholders in research initiatives can occur at different phases of a project's implementation and to different extents. Table 3 describes the phases and events of the BMZ-funded project that were critical to identifying the stakeholders and to the consultative decision-making that occurred during the international and local implementation of research activities. A limited number of the eventual stakeholders participated in the actual development of the project proposal and, in fact, the intended project beneficiaries were not approached during this phase. Although Table 3 shows wide participation of stakeholders, especially in the planning workshops, in reality, other than local progress meetings, personnel of the international and national research institutions were the main participants.

A stepped approach was adopted to identify and reach more stakeholders through a series of planning and consultation meetings that took place after the project had started. Stakeholders that were identified during one meeting were invited to attend the following consultation rounds. At subsequent meetings, the objectives and the proposed activities of the project were discussed again. This pattern parallels the reiterative process described in Figure 1.

The project was first conceptualized when the International Plant Genetic Resources Institute (IPGRI) and several forest research institutions in Germany recognized the problem

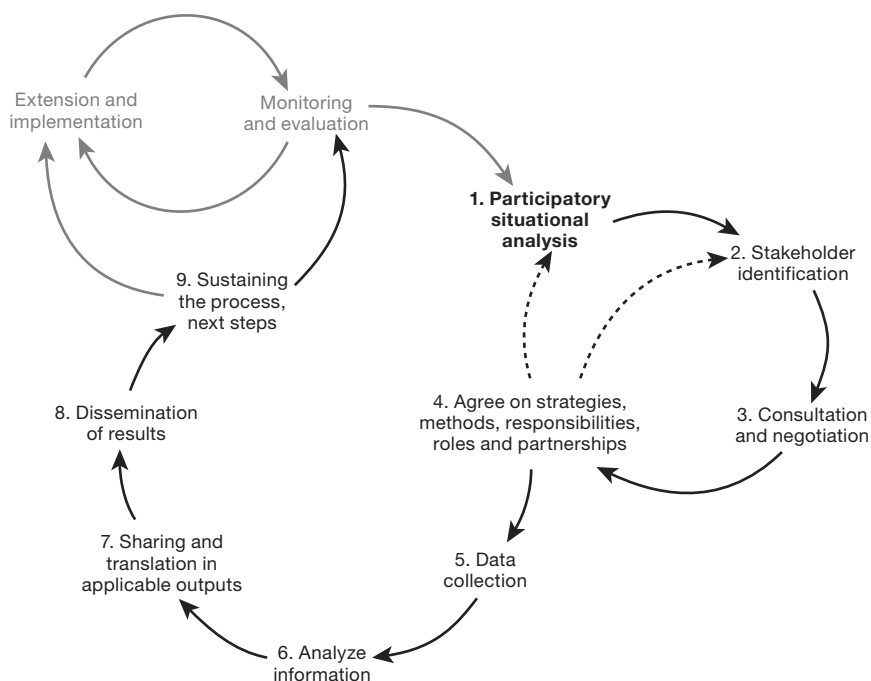


Figure 1. Project phases in 'ideal' participatory research and development projects, with possible reiterative routes in the initial phases of the project.

Table 3. Principal decision-making events, their objectives and participants.

Date/period	Event/activity	Participants ¹									Principal objectives ²	Initiators/ organizers
		a	b	c	d	e	f	g	h	i		
1998	Project proposal development	X									A, B (national level), D	a
June 1999	Initial planning workshop	X	X	X	X	X			X	X	A, B (regional level), C, D	a
July 1999 – Feb 2000	6-month preparation period		X			X	X				A, B (local level)	b
Feb/March 2000	Local planning workshops ³	X	X	X	X	X	X	X	X		A, C, D	a, b
October 2000	Project coordination meeting	X	X								C, D, E	a
Several, during implementation phase (June 2000 – April 2003)	Local progress meetings ³		X	X	X		X	X			D, E, F	b
	Local synthesis workshops	X	X	X	X	X	X	X	X		E, F	a, b
April/May 2003	Final project workshop	X	X						X		F	a, b

¹Principal stakeholder groups and partners:

a = International research institutions and national research institutions from developing countries

b = National research institutions from participating countries

c = Environmental NGOs

d = National and/or regional policy-makers and implementing agencies

e = National and/or regional NGOs

f = Local or locally active civil society organizations (NGOs, farmers associations)

g = Representatives of local communities

h = Resource persons from different research institutions

i = Private sector representatives

² A = Situation analysis, B = Stakeholder identification, C = Consultation, D = Decisions on changes in project framework, objectives and activities, E = Sharing information on progress and/or decisions, F = Formulation of results, follow-up discussion

³ Objectives, set-up and participation differed among sites, and per event.

of continuing deforestation and genetic erosion of FGRs worldwide, but in particular in Brazil and Argentina, and an opportunity presented itself to work together with stakeholders in these countries to contribute to its resolution. Time and financial constraints hindered a full stakeholder analysis at this point (as is frequently the case in project planning in research on conservation and sustainable use of natural resources, forestry and agriculture). The proposal submitted to the donor contained a general evaluation of the problem based on limited consultations between the international institutions and the national research partners in Brazil and Argentina.

We felt that while a full and early local consultation would have been ideal, a stepped process that progressively involved more local stakeholders as the research activities began could be a successful alternative. Having adopted this strategy, we made efforts to ensure that the formal project procedures were flexible enough to allow for significant changes in the original work plan as consultations with local communities and other stakeholders revealed their inputs on and concerns about the project.

2.4 Identification of stakeholders and degrees of involvement

An essential step in adopting PAs is the early identification of stakeholders. In general terms, stakeholders are those people, groups or organizations who have an interest in an issue. All people who are influencing a decision, can influence it, or are affected by it, are potentially concerned (Earth Summit 2002). As such, people who are intermediaries and without direct interests themselves can also be stakeholders (Brown 1998).

As noted by Enters (2000) and Cramb *et al.* (2000), assumptions on whether communities are stable or homogeneous, and on the interest, commitment, knowledge or skills rural people possess, need to be carefully reviewed. Normally, communities are rarely harmonious groupings with single agendas on different subjects – development for instance – but are groups of stakeholders with dissimilar needs, priorities, and capabilities (Leach *et al.* 1997). Identifying these groups and their agendas can be difficult. An equally important challenge is to determine who (can) represent(s) a community. Communities may have various leaders representing a variety of community factions, or leaders with distinct roles and responsibilities. Moreover, one organization may claim to represent a community, but upon closer examination this organization's leadership may be found to be unacceptable to the (whole) community. Similar observations can be made about other stakeholder groups like 'the government', 'environmental groups', or 'scientists'. For example, even when the objectives of environmental organizations are similar, they can hold conflicting views on how to approach a problem. And when competition for funds comes into play, factors that determine relationships and the distribution of power within and among stakeholder groups can further disrupt progress.

Furthermore, stakeholder representatives may not be willing or able to maintain effective two-way or multiple-channel communication with their constituencies about the participatory process. If gaps develop between the expectations of stakeholders and the stances taken by their representatives, this may result in decisions or outcomes that lack stakeholder backing, with consequent problems when implementation is expected to occur (Buchy *et al.* 2000).

Degrees of involvement of project stakeholders can differ considerably. The most obvious way to evaluate the success of the PA adopted in any given project is to assess the levels and types of participation of the (intended) local beneficiaries, the organizations that are engaged in project implementation, and those organizations or beneficiaries that stand to gain from the outcomes. According to a typology of participation developed by Pretty (1994; 1995), two primary forms of participation are distinguishable (Table 4). The first type of participation, and the one most often found in target-oriented PAs, is relatively passive, where people participate by being told what is happening. The exchange is typically one-sided, aiming to inform the public about activities or outcomes, or sometimes to educate them, and thereby to give legitimacy to the project and to further acceptance of its products.

The second type of participation, and the one usually found in process-oriented PAs, might be called 'interactive participation'. This is where the public actively participates in joint analyses and decisions that lead to locally formulated action plans, and where joint learning capacity is enhanced. Here, communication is typically two-way, with significant stakeholder deliberation, and with the objectives of educating participants about issues and gathering data, including the knowledge of participants, and with experts and other stakeholders developing the best options together. In this type of PA, participants are involved in a continuous exchange, and their ideas are incorporated to the greatest extent possible into the project design and its outcomes.

A range of power-sharing possibilities exists between Pretty's two types of participation (Buchy *et al.* 2000), which are not mutually exclusive. Furthermore, the level and form of participation can change over time and it may differ according to the activity in which people are involved or the topic that is under discussion. This may result in dissimilar types of participation following each other in a project cycle. For instance, when a project enters an implementation phase (e.g., to experimentation from data collection) participants and their various relationships may change as new partners join the project and different PAs are employed.

Active participation of the direct stakeholders at early stages of problem definition and research priority-setting has important advantages in PGR or FGR development projects. Benefits include early definition of what technologies or behavioural changes users are likely to accept, as well as the opportunity to adapt prototype technologies earlier in the project in ways that will meet the needs and preferences of both the beneficiaries and the expert outsiders (Systemwide Programme on Participatory Research and Gender Analysis 1997).

Table 4. Typology of participation.

Participation type	Characteristics
Passive participation	Most people participate by being informed about decisions, what is happening, or what has already happened. The information being shared originates from external professionals. An example is when research is designed, planned and carried out by external professionals. Most stakeholders are presented a ready-made solution or, at best, several to choose from
Participation by consultation	Most people participate by being consulted or by answering questions posed by external professionals. The process does not concede any share in decision-making and professionals are under no obligation to accept other viewpoints. An example is when external professionals use questionnaires to obtain knowledge from local farmers
Reimbursed participation	People participate in return for food, cash or other material incentives. Local participants have no stake in prolonging technologies or practices when the incentives end. An example is when local forest dwellers are employed to carry out or help in fieldwork
Interactive participation	People participate in joint analyses, action plan development, and formation or strengthening of relevant local institutions. Methods are used that seek multiple perspectives and local groups help determine how available resources are used. An example is when researchers develop an idea for a project and invite local farmers to participate in the design, planning and implementation of all activities
Self-mobilization	People participate by independently taking the initiative in order to change the status quo. They initiate contacts with external institutions to gain resources and technical advice. They retain control over resource use and project outcomes

Source: adapted from Pretty 1995 (<http://www2.essex.ac.uk/ces/ResearchProgrammes/pa&caover4.htm>)

As discussed above, in some research projects or project phases like those involving highly technical assessments of genetic diversity, local participation may not be useful or necessary. In projects like these, local community participation will usually be passive and/or limited to remunerated fieldwork, with one-way communication occurring in the form of consultation. Nevertheless, a continuous exchange of information is desirable because even when stakeholders cannot provide effective inputs, their continued interest in the project will be needed to ensure their continued support for it.

Many stakeholders were involved in the multidisciplinary, multisite BMZ-funded project. Because of this, we felt the need to distinguish between participation and partnership. We determined that an institution or a group of people were partners when they had project responsibilities that variously influenced elements or processes of the project. We determined that people or institutions were participants when they were involved in using or managing forest resources and thereby had interactions with the professional researchers. Thus, we determined that the intended project beneficiaries were principally participants who had an important stake in the outcome. While Buchy *et al.* (2000) state that participation is all about partnerships, there are types of participation that do not influence or become responsible for project implementation, but rather receive information and other products (Table 4 p. 231). In this regard, collaboration among stakeholders is largely based on power relations. When decisions are made without the beneficiaries, who are merely assigned the role of implementing and evaluating results, power inequalities between participants and partners are maintained or even enforced (Waisbord 2001).

To understand stakeholder interactions in participatory research, we examined the partnerships and participants in this project, including the division of responsibilities, the mechanisms for decision-making, the relations within and among groups of stakeholders, and the extent to which the stated beneficiaries were included and benefited from the project.

Lessons learned from the BMZ-funded project: identification and involvement of stakeholders

Because project activities were proposed at four sites in two countries, stakeholder identification was initiated by national research institutions (Escola Superior de Agricultura Luiz de Queiroz, Universidade de São Paulo – ESALQ; Universidade Federal do Paraná – UFPR; Instituto Nacional de Tecnologia Agropecuária – INTA), which then became responsible for seeking and facilitating community involvement throughout the research phase of the project. The national organizations developed relationships with local stakeholder groups and assisted them in becoming involved in the organization and completion of fieldwork. In turn, local stakeholders facilitated access for national organization researchers to the field sites.

In Pontal, Acre and Paraná (Brazil) stakeholders were important interest groups or organizations that represented professions such as the National Rubber-Tappers Movement (CNS), the Cooperative of the Landless Movement in Pontal (MST), the settlers' cooperative COCAMP (Cooperativa de Comercialização e Prestação de Serviços dos Assentados de Reforma Agrária do Pontal do Paranapanema), and rural syndicates in Paraná. These bodies acted as intermediaries between the projects and the representatives of local communities. At the project site in Argentina, this role was played by INTA's section for rural development.

This stepped approach for developing relationships seemed to work well. We were confronted with the task of organizing projects in two countries and at many sites and involving several scientific disciplines and a myriad of rural communities and local interest groups, and all this was to be implemented across a range of environmental and socioeconomic contexts.

From the outset, the BMZ project had a strong research orientation. The indigenous communities in Acre (Brazil) and Argentina had already collaborated with other research projects and claimed to have seen few tangible results (ETC, ESALQ and IPGRI in 2002, and ETC, INTA and IPGRI in 2002). Small development initiatives were then incorporated into

the projects in three of the four sites (Argentina, Pontal and Paraná). This triggered further development efforts at two sites, with additional funding from other sources later obtained to continue these activities.

Appropriate representation of local stakeholders is a fundamental issue in projects adopting PAs, and it is important that mechanisms are in place to ensure the identification of all relevant stakeholders, followed by the selection of partners and their respective representatives. In practice, initial contact is usually made with well-known institutions and organizations in the rural communities, and community members who are not aligned with such organizations can be easily overlooked simply because they are unaware of the project during the planning phase or, even if aware of the project, they may not come forwards to express their interest in it. Community politics are dynamic and it might also be the case that subgroups and both formal and informal leaders have different views, even within the same organization. Thus the selection of representative members from the rural communities became an important undertaking requiring considerable care and discernment.

In Pontal, COCAMP played a strong intermediary role between the project and the rural communities. Even though COCAMP leadership was not recognized by all the farmers, the project did manage to interact with a relatively large fraction of the communities and even with those not directly involved in the project. This facilitated the dissemination and the use of research outputs in other communities. The same happened at one of the Acre field sites where a large part of the rubber-tapper community became involved in the project through the efforts of the CNS and the local rubber-tapper association. In contrast, the lack of any clear leadership structure in the agricultural settlements of Acre made it difficult to establish good communication channels with local settlers.

In practice, at all sites, most collaboration was between a small number of individuals, families or representatives appointed by local leadership and local field coordinators and representatives of national or international institutions and organizations. Through the assumption that local participants would disseminate information and ideas to the rest of their communities, either directly or through the body or organization they represented, the project more-or-less treated communities as homogeneous groups. No substantive assessment of participatory processes occurred and, as a consequence, we are unable to say whether all interested and/or relevant community members, especially marginalized groups, were reached.

Lessons learned from the BMZ-funded project: participation as a means of building partnerships

Patterns of information exchange

The BMZ-funded project used participatory tools to gather information on the uses of FGRs and on their place in local economies and livelihoods. Participation of local people in this phase of research was mostly passive and did not influence either the definition or analysis of problems, or the identification of potential solutions. An assessment carried out during the synthesis workshop in Pontal on the quality of the partnerships that were established during the project confirmed that researchers from national institutions played a dominant role in organizing and conducting research activities and that exchanges of information with beneficiaries was closest to what we earlier called ‘transfer of technology’ (Table 2 p. 227). While project aims were to enhance the understanding of rural people’s role in forest management and use, the project was not sufficiently oriented towards incorporating indigenous knowledge and perspectives into the design of the activities, nor into final outcomes.

Creating ownership and partnerships through consultation

Participatory processes were also adopted in order to introduce flexibility into the project’s work plan. Table 3 (p. 229) illustrates the extensive consultations that took place during project implementation in order to review objectives and to prioritize activities. Consultations occurred through general and local planning meetings; the latter turned out

to be especially important in stimulating the participation of local stakeholders in decision-making. Still, the national research coordinators and partners maintained the leading roles throughout the life of the project. On several occasions, national research partners and local stakeholders proposed significant changes to the original work plan. However, because there was an overriding tendency to try to fulfil donor expectations and to avoid substantive changes in work plans, only slight modifications were introduced. This was at odds with what Davis (1996) identified as a fundamental element of a true participatory approach, where “The outcome of consultation should not be predetermined.” Nevertheless, as we will discuss in more detail below, we did introduce several development-oriented activities at the request of participants of the local workshops organized at each study site in the course of the project implementation.

Although we did not manage to achieve a fully participatory approach, local workshops helped to establish a consultative-type involvement of and interactions between researchers and local stakeholders. The fact that participation remained largely consultative was somewhat surprising, as some local partners such as CNS and COCAMP normally aim to establish only interactive or self-mobilizing partnerships (Table 4 p. 231). Their more passive role in the project reflected a disjunction between the scientific interests of researchers and the development perspective of these local organizations.

At the Paraná research site, an interactive network was established among the research organizations involved, but there was little participation from local communities. In contrast, in Argentina, a consultative relationship was established between researchers and local people, leading to some limited ownership of the project by indigenous communities.

The assumption that researchers could facilitate and conduct participatory research themselves was important to project design. But the reality of stakeholder interactions and project outcomes indicated that this part of the project needed strengthening. For example, local planning workshops led to only very limited participation of a consultative and/or collaborative nature by local communities in project implementation phases. Consequently, the commitment of local communities and other local stakeholders to research and to other outcomes was weak.

In this regard, it is important to appreciate that research partners were identified on the basis of their ability to conduct FGR research and not on their potential to work in a participatory manner. The design and implementation of PAs requires careful preparation, and these are not part of the background of most biologists. As a consequence, we learned that when partners in a research project agree to apply PAs, their capacity to conduct research under these terms must be assessed beforehand. And possessing this skill, or being willing to acquire it, should be a condition of the acceptance of new partners. We now realize that PA capacity building should have been incorporated into the overall project design.

Local progress workshops

Significant stakeholder participation in the inception of a project does not guarantee stakeholder engagement in later phases, nor does a lack of initial engagement preclude later active involvement. Both of these scenarios are dependent, in part at least, on the ability of stakeholders to express their views and to influence the course of a project. To secure continuous engagement of local stakeholders, project meetings were organized at different times at each of the BMZ-funded project sites, with the objective being to share and discuss research progress and findings. These gatherings did not produce significant changes in the work plan nor did they trigger engagement with additional stakeholders. Nevertheless, they did stimulate follow-up activities in Argentina and Pontal (Brazil).

Local synthesis workshops (research results and stakeholder participation)

During July and August 2002, synthesis workshops were organized at each project site. The first part of these workshops was dedicated to an exchange of research findings and to attempts to gather results into a common framework. In the second

part, local stakeholders joined the meetings, and findings from genetic, ecological and socioeconomic research were presented with recommendations made to guide implementation of conservation strategies for FGRs. The goal of this second part was for the participants to discuss the research results, translate them into practical recommendations, and agree on next steps.

At all but one site, difficulties emerged in creating a framework into which research findings from the several disciplines could be integrated. The project workshop in Argentina was an exception, thanks to the use of a systems approach that enabled the data from different disciplines to be combined into a single model (described in Chapter 10). This, in turn, permitted translation of research results into conservation strategies, management practices and public policies. In this context, it is fair to say that we found that PAs did not play an essential role in creating this interdisciplinary or systems approach. It was instead the result of initiatives by the researchers themselves.

Final synthesis workshops

Building solid links and a continuous exchange of information among the project sites was a difficult challenge that was only partially achieved. Ultimately, the project functioned more like a 'collection of case studies' than as an integrated network of research field sites. During the final synthesis workshop, emphasis was placed on those outputs that contributed to achieving general project objectives (IPGRI 2003) and it was not structured in a way that allowed broad participation. Research coordinators, some research partners and resource persons were invited, but no local stakeholders were invited to attend this gathering. Consequently, the consultative nature of the initial project meeting was not paralleled by a final sharing of outputs at the project's conclusion. This choice was largely dictated by funding limitations, and it thus illustrates the importance of adequate budgeting for stakeholders' participation throughout all phases of a project of this kind.

Development activities

One of the critiques advanced by representatives of local research partners in the planning and progress meetings, and during the synthesis workshops, was the lack of tangible results that would benefit local stakeholders. This is a common criticism of projects that are primarily research-oriented and occur among local communities, that is, local stakeholders may expect concrete answers to immediate problems that in many cases cannot be provided by the research results. Solving immediate local problems was neither a primary objective of the BMZ-funded project nor was it realistically possible. Researchers must be transparent about this, or project acceptance by local participants will be even less likely. If research projects like this one are expected to respond to local and other stakeholders' demands, much more operational flexibility will be needed along with the financial resources that permit inclusion of such additional activities. Taking a more opportunistic approach in project planning and implementation, where activities of more interest to local people can also be funded, will garner support for medium- to long-term research projects while enhancing the chances of solving some urgent local problems related to resource sustainability.

To a certain extent, the BMZ-funded project adopted a more flexible approach following demands from local stakeholders at the planning meetings. Among the more visible outcomes were training opportunities for local stakeholders, organized at three of the project sites in Pontal, Paraná (Brazil) and Argentina. For example, in Pontal an agro-forestry experiment with demonstration plots was added, as suggested by local community members during the regional workshop. The site and the species for these agro-forestry trials were selected locally through participatory meetings with representatives from 12 communities, and with the active engagement of COCAMP, which later became involved as a leading partner in the planning and preparation of a new project proposal that built on the results of this initiative.

To address local demands, a herbarium of medicinal plants was developed in Paraná by a local NGO in collaboration with a rural community. This initiative focused attention

on the BMZ-funded project from a community that was only marginally touched by the original project plan.

Similarly, in Argentina, indigenous Mapuche communities requested that more development-oriented activities be included in the work plan, and a modest tree nursery was subsequently added. While this project was only partially implemented owing to funding constraints, it became an important element in building collaboration between the local communities and the project managers, and regular meetings between researchers and community representatives eventually led to a much more constructive dialogue.

In conclusion, the most successful collaborations, especially those in Argentina and Pontal, occurred in contexts where relationships started out being consultative between project staff and local stakeholders but broadened when activities were developed that had more visible and immediate benefits for local communities.

2.5 Participatory approaches in a context of adaptive management of FGRs

Participation of local people should be seen as a means of linking academic biological research with research on local resource management and use practices, as well as facilitating the incorporation of the needs and perspectives of local stakeholders in the research activities themselves, with the aim of developing better management policies and practices. An emerging perspective that stresses the use of PAs in ecosystem management and policy development within a context involving people is called 'adaptive management'.

Adaptive management is a concept that has been developed over the past three decades by ecologists (e.g., Holling 1986; 1995; Lee 1993) engaged in the management of large ecosystems like watersheds, lakes or forests and of a range of natural resources such as water, forests, grasslands or fish. In this perspective, natural resource management and conservation policies are thought to be effective only when planned and implemented within a framework of partnerships. Links between policy, management and local stakeholders are central to these partnerships (Gunderson *et al.* 1995; Berkes *et al.* 1998). Through these links and appropriate feedback mechanisms, stakeholders can learn from each other's experiences.

The role of research in adaptive management is to provide and strengthen feedback and learning among policy, management and local people (Figure 2). Research provides the ability to monitor management practices, assess the effects of policies, evaluate involvement of local stakeholders, generate essential information about ecological and social processes, and translate new knowledge into new or revised management options. Research also makes policy and management organizations more responsive to socioeconomic variables and ecological change (Gunderson *et al.* 1995; de Boef 2000).

In the following section, an adaptive management perspective is used to analyze those parts of the BMZ-funded project where policy, management and local people were linked through research. We particularly wish to assess the degree to which the BMZ-funded project's PA yielded partnerships between local people and research, policy and management organizations, and facilitated capacity building for shared learning. We will look specifically at the following questions:

- Did the project achieve participation of local people and other intended end-users?
- Were local people approached as key stakeholders with their own contributions to conservation and given legitimate consideration within an adaptive management perspective?
- In what ways did the research provide inputs for policy-makers, management institutions and local people?
- Did research contribute to stakeholders joining forces in common action and learning?
- Did the project lead to new initiatives by the target groups, or in any way stimulate target groups to want to continue any part of the project?

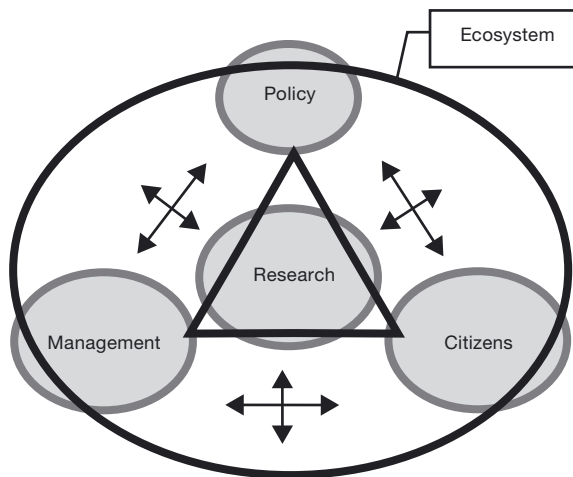


Figure 2. Triangle of adaptive ecosystem management. Management affects stakeholders that variously manage resources, use resources and services from ecosystems, and benefit economically from these activities. Policy organizations create institutional frameworks in which management organizations operate. Citizens are the (local) stakeholders that depend on the natural resources, are affected by management and policy decisions concerning these resources, and/or affect them through their own actions. Research provides mechanisms for feedback and learning. Source: de Boef 2000.

Lessons learned from the BMZ-funded project: an adaptive management perspective

Whether we described them as rural communities or local resident organizations, like in Paraná, Acre and Argentina, or as local civil organizations such as MST/COCAMP in Pontal or CNS in Acre, the local people at all the research sites were viewed by the BMZ-funded project as key players in FGR conservation and management. Nevertheless, in practical terms, their degree of FGR conservation and use varied a good deal. For instance, rubber-tappers were highly dependent on forest products, while forest settlers in Pontal were mainly dependent on agriculture and livestock grazing. Consequently, in their efforts to involve them in the BMZ-funded project, researchers approached local people and organizations in different ways. Below, we describe local contributions to research at the four principal study sites. We also evaluate the extent to which research activities were able to link partnership-building with management and policy, and whether research provided outcomes leading to adaptive FGR management.

Argentina

The project in Argentina (Chapters 6 and 10) focused on the Mapuche people and their use and management of araucaria forests. Systems analysis research demonstrated a fundamental role for the Mapuche in araucaria use and management. Consequently, we concluded that any policy and management recommendations we might make would need to apply both to araucaria forest populations and to the manner in which they were managed by the indigenous communities. Nevertheless, Mapuche participation in the project was largely confined to planning workshops and various local meetings that occurred between research partners and community leaders. The socioeconomic surveys, using participatory tools and conducted among Mapuche, provided valuable insights into

their role in araucaria management, and when this aspect of forest use and management was discussed at meetings, community members showed active interest, much as they did during discussions on tree nurseries. This consultative approach resulted in the initiation of a rural development project, the main activities of which were determined by the members of the two Mapuche communities who thus played a proactive role in this phase of the project cycle from problem identification to project implementation.

At the outset, the project in Argentina treated regional policy-making and management agencies as key stakeholders. Even before the project was started, several regional stakeholders were already organized into an informal network involving araucaria conservation. As a result of the BMZ project, the management staff at Lanin National Park decentralized administrative functions and increased the number of local staff assigned to conservation. The administration of the park and the Neuquén provincial authorities also agreed to coordinate their respective work within the national park and jointly to address environmental and social problems of the Aigo (Mapuche) indigenous community living in the area. Moreover, the Aigo community decided to send a representative to work with the park/province coordinating team. The project thus contributed to the formation of a stakeholder network of national and local partners that was involved in making decisions on policy, management and use. As such, an adaptive management framework was created during the implementation of that project (IPGRI 2003).

Paraná (Brazil)

While the projects at the other study sites focused on interactions between local communities and target species and ecosystems, the Paraná project focused solely on araucaria ecosystem management (Chapter 7). The primary event was an early consultative meeting organized by the local research partners. Representatives from state level research institutions and policy-making and implementing agencies and from the private sector (forestry and agriculture) discussed problems associated with araucaria conservation and use. No local communities participated in this or other meetings. As a consequence, stakeholders were defined as the policy-makers, the implementing agencies and the private forestry sector, and representatives of these organizations operated as a network that included various state and federal research partners.

This research framework resulted in strong scientific outcomes that were translated into concrete guidelines for conservation and sustainable use. Recommendations were also made in the areas of policy change and follow-up research. Even though the project did not specifically provide for mechanisms that would further the adoption of guidelines and recommendations, we felt that the partnerships that were created during the project boded well for the implementation of some recommendations once the project was concluded. In contrast, while some activities with local communities did occur, there was little to no involvement of local communities as the project unfolded. The Paraná project thus only partially achieved adaptive management status for araucaria. While it focused on araucaria conservation, management and use, it involved mainly agency and industry representatives.

Pontal (Brazil)

Local participation in the BMZ-funded project at Pontal mostly occurred through the organization COCAMP. We thus expected significant involvement of rural people and their community organizations in FGR management and conservation. The settler communities were approached at the outset as potential key players in the conservation and use of forest products. Representatives of these communities, and especially COCAMP, embraced the conservation and sustainable use notions advocated by the researchers, and they especially showed interest in an agro-forestry approach. Unlike the Mapuche people in Argentina and the rubber-tappers in Acre, the Pontal settlers were not directly dependent on the forest and its products for their livelihoods.

Because of its genetic and ecological orientations, the research project provided scientific information that was important to FGR conservation and it highlighted the critical

conservation functions of individual forest fragments in the areas inhabited by the settlers. However, participation of community representatives in actual research was limited, though some did occur through contracts and passive participation. Participation of local stakeholders other than COCAMP was consultative in the initial phase, through the planning workshops, and became more passive during the planning and synthesis workshops.

One consequence of this approach was that the settlers and their local organizations expressed only modest ownership over project results. Tangible and constructive outcomes for the local communities were limited, and it is questionable if the project contributed to an improvement in local FGR management. However, even though implementation of conservation outcomes and participation in research activities were limited, the experimental agro-forestry activities did involve many local people in a participatory manner. As a consequence, COCAMP and other community members expressed much stronger interest in sustainable use of FGRs. Eventually, this led to the development of several new initiatives and project proposals that supported the cultivation and use of medicinal plants and the development of sylvipastoral systems. In one of these initiatives, COCAMP took the lead with collaboration from ESALQ and several other policy-making and implementing agencies as contributors.

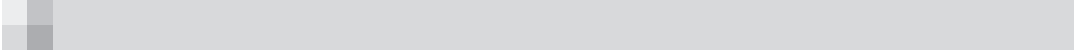
Thus, while the participation of rural communities in the research activities themselves was limited, the introduction of a project component that corresponded with local needs and interests contributed to wider acceptance of the project and, we hope, its results. Because of the positive political climate that developed among the key players in the region, and especially with COCAMP, the state park authorities and the state land management agency, the project assisted in creating a more proactive attitude among local stakeholders towards conservation and sustainable use issues. The project thus contributed to creating and shaping local partnerships for decision-making and action and in so doing substantially moved Pontal towards more effective adaptive management of the park and its surrounding forest fragments.

Acre (Brazil)

At the Acre study site the project worked with two communities – the rubber-tappers and the agricultural settlers – and with their associated land-use types. The links between FGRs and rubber-tappers were more evident at Acre than at any other of the project sites as they depend largely on forest products (such as rubber and *pará* nuts) for their income. Links between the agricultural settlers and the forest were similar to those of the Pontal settlers, where few direct uses existed other than during the initial phase of settlement when the forest was used for hunting and construction materials. The degree of organization of the rubber-tapper communities appeared strong at the outset. Their community leaders participated actively in project discussions and contributed to field research, though usually through contracts. Organization of the agricultural settlers was much weaker, perhaps owing to the fact that the settlements were recent. Settler participation was largely limited to one family.

The dissimilar economic strategies of the rubber-tappers and of the newly established settlers were the main issues addressed by the socioeconomic research, which maintained a focus on the different effects of subsistence and income-generating activities on long-term FGR conservation. The focus of the socioeconomic research was descriptive rather than problem solving. That, and the limited use of participatory tools in the research and in the contractual and consultative interactions between researchers and members of both communities, resulted in rather passive involvement of the two communities in the project. As such, during the socioeconomic, genetic and ecological studies, only limited contributions were made by the project in addressing the resource-use conflicts that were identified between the rubber-tappers and the settlers.

The partnership that we hoped would develop between stakeholders involved in research (Federal University of Acre, EMBRAPA - Empresa Brasileira de Pesquisa Agropecuária, Acre) and policy and management (State Agency for Environmental Management) did not occur. This was due mostly to the fact that an external research organization (ESALQ) coordinated and implemented the research through a small group of community leaders



and interested individuals. This partly contributed to an exclusion of other institutions and prevented the formation of partnerships with other important stakeholders. Consequently, the good potential we thought we had in Acre for partnership-building and for adaptive FGR management was not achieved. When we consider that forest management and 'extractivism' are key political concerns in Acre State, we were forced to conclude that the project made only modest contributions there.

In drawing conclusions from the Acre project site it should be made clear that many of the organizations that joined in the BMZ-funded project there were also involved in other Brazilian forest management and conservation projects and activities. The Acre BMZ-funded subproject was small compared with the many other continuing projects in Brazil that were tackling similar issues, and this may have limited the interest of local policy-makers and research institutions in supporting this research initiative. Consequently, the Acre subproject was less successful in developing partnerships within an adaptive FGR management context.

3. Concluding remarks

Capacity building in participatory research

We have seen that project researchers were motivated to involve local people in their activities. However, we found that it was incorrect to assume that researchers had the skills to achieve stakeholder participation even when local people were interested in becoming involved. We also found that the inclusion of socioeconomic research does not guarantee the success of a participatory approach (PA).

Bearing in mind that one of the guiding principles of the project was to use a PA, and that local people were viewed as stakeholders with important contributions of their own to make toward conservation and sustainable management, we still found a striking gap between the actual performance of the FGR research groups and what we expected them to produce using PAs. Thus, one project design oversight was that the ecological, genetic and socioeconomic researchers, although highly qualified in their respective scientific disciplines, should also have been trained to conduct participatory research, or that experts in participatory methods should have been included in the research teams from the outset of the project. This is an important lesson for donors and international agencies when they support and initiate projects like the one described in this book. A participatory project is inadequate unless the researchers involved have the skills or the support they need to engage in participatory research.

Power relations and representation

The first phase of the BMZ-funded project included a broad effort to identify and consult with potential stakeholders. However, many of the people, organizations and institutions identified ultimately played little or no role in implementation phases. This was expected as interest, capacity and (financial and technical) resources will usually determine who finally participates. However, we should also recognize the importance of power relations. During the many meetings at different places a common complaint heard involved money and responsibilities and especially that both were unevenly distributed among partners. Relations of differential power were clearly at issue. At the level of the study sites, power in terms of research and financial responsibility was also unequally distributed. At all sites, research supervision and resource allocation were strictly coordinated by the lead national research partners (in Brazil: ESALQ in Pontal and Acre, UFP in Paran ; INTA in Argentina) and this resulted in limited manoeuvring room for the locally stationed junior researchers, research partners, and community representatives who supported the research activities. This structure did little to promote project ownership by other stakeholders and resulted in only very modest partnership-building, both of which are required for collective learning and adaptive FGR management.

Power relations also influenced who could or wanted to represent the indigenous and settler communities and the professional organizations. We saw that participation of local

people was usually limited to a few individuals, often through civil society organizations (Pontal, Acre) or community bodies (Argentina, Paraná). The project did not provide mechanisms to facilitate or evaluate information flow and consultation between these individuals and the communities they came from or represented. We implicitly assumed that individuals from or representing communities would reflect the needs and the opinions of those communities, including marginalized groups. This is an assumption we now believe was unjustified, given what we now know about the complexity of community and community–environment relations. The fact that we cannot say which community members benefited from the various projects underscores the importance of monitoring and evaluating power relations and interactions among community members and their representatives. This would have helped us to evaluate the success of the PA and thus to draw lessons for future projects. Just as importantly, it would have allowed for timely adjustments to the project design and its implementation in mid-course.

Time frames

An observation frequently made by local project coordinators was the difficulty in completing projects within established time frames. The report of one of four final regional workshops carried out in each of the project sites (ETC *et al.* 2002) phrases it as follows: “The period of these types of participatory research projects is very short to achieve a real involvement and participation of the communities. It is necessary that the people can understand the importance of the social and natural environment research. This is a complex process that normally lasts decades.”

As the initial focus of the project was on participatory research that was geared towards obtaining specific ecological, genetic and socioeconomic information, the planned time frame of three years should have been sufficient. Given this research focus and the limited effective outcomes that resulted, it is not surprising that contributions to local FGR management were partial at best. Because local participation was generally low, specific activities tailored more to the immediate needs of the local communities were initiated. This programmatic change proved to be an essential step in gaining support or, in some cases, changes in attitude towards the projects. At Pontal and in Argentina, the BMZ project helped to initiate community activities developed by local stakeholders.

The project now completed could be seen as the research phase of a larger undertaking (Figure 1 p. 228), and it could find its impact enormously enhanced if a complementary development programme was initiated soon to build on its achievements. Such a programme would make it much more likely that the BMZ project would lead to lasting changes in the management and use of FGRs at the respective research sites. This assumption underlines the importance of planning participatory research and development programmes that also address medium- and long-term goals linked to the essential first steps of gathering and analysing information through research. However, this conclusion should not cause us to underestimate projects of limited time and scope for as we saw, if well-conceived, they can also play important roles in winning the confidence and the support of local people.

Partnership formation

Research boundaries at the BMZ-funded project study sites were defined using geographic, biological, political and socioeconomic criteria. The sites, the species and the research and development partners were identified during consultative meetings and were thereby to a certain degree ‘constructed’ at that time. To some extent, these sites have now led to the formation of additional partnerships for managing the FGRs that they contain. A shared realization that conservation and sustainable use are emergent and important properties of these sites has now begun to bind some of the local stakeholders with other project partners.

Even though it operated within a range of sites and settings and it adopted several strategies to achieve participation, the BMZ-funded project contributed to the formation of partnerships among researchers and rural organizations with the common objective of improving FGR management and related policies. In this context, more guidance and training for biological scientists in participatory research and development design

will be required in order to create or shape future partnerships in sustainable FGR use and management involving rural people. Only when development outputs are linked to improvements in rural livelihoods will true progress beyond genetic, ecological and socioeconomic research be achieved.

References

- Almekinders, C. and W. de Boef (eds.). 2000. Encouraging diversity. The conservation and development of plant genetic resources. Intermediate Technology Publications, London, UK.
- Ashby, J. 2003. Uniting science and participation in the process of innovation – research for development. Pp. 1-15 *in* Managing Natural Resources for Sustainable Livelihoods: Uniting Science and Participation (B. Pound, S. Snapp, C. McDougall and A. Braun, eds.). International Development Research Centre (IDRC) and Earthscan Books, London, UK.
- Barrance, A.J. 1996. Social impact considerations in protected area planning. MSc Thesis, University of Wales, Aberystwyth, UK.
- Berkes, F., C. Folke and J. Colding (eds.). 1998. Linking Social and Ecological Systems. Management Practices and Social Mechanisms. Cambridge University Press. Cambridge, UK.
- Braidwood, R.J. 1967. Prehistoric Men. Scott, Foresman and Co, Glenview, USA.
- Brown, K. 1998. The political ecology of biodiversity, conservation and development in Nepal's Terai: Confused meanings, means and ends. *Ecol. Econ.* 24:73-87.
- Buchy, M., H. Ross and W. Proctor. 2000. Enhancing the information base on participatory approaches in Australian natural resource management. Commissioned research under Land and Water Australia's Social and Institutional Research Programme. http://www.lwa.gov.au/downloads/final_reports/ANU21.pdf
- Cramb, R.A., J.N.M. Garcia, R.V. Gerrits and G.C. Saguiguit. 2000. Conservation farming projects in the Philippine uplands: rhetoric and reality. *World Dev.* 28(5):911-927.
- Davis, G. 1996. Consultation, Public Participation and the Integration of Multiple Interests into Policy Making. Organization for Economic Cooperation and Development (OECD), Paris, France.
- de Boef, W.S. 2000. Tales of the unpredictable. Learning about institutional frameworks that support farmer management of agro-biodiversity. PhD thesis. Wageningen University. Wageningen, The Netherlands.
- Earth Summit. 2002. <http://www.earthsummit2002.org/ic/>
- Enters, T. 2000. Rethinking stakeholder involvement in biodiversity conservation projects. Pp. 263-273 *in* Forest Conservation Genetics: Principles and Practice (Y.D. Boshier and T. Boyle, eds.). CSIRO publishing, Collingwood, Australia and CAB International, Wallingford, UK.
- ETC, ESALQ and IPGRI (ETC, Escola Superior Agricultura Luiz de Querioz and the International Plant Genetic Resources Institute). 2002. Relatório da Reunião de Trabalho 25-26 de julho de 2002, Rio Branco-AC, Brasil. Projeto Acre. Conservação, Manejo e Uso Sustentável de Recursos Genéticos Florestais com Referência ao Brasil e Argentina. ETC The Netherlands, ESALQ Piracicaba, Brazil, IPGRI Rome, Italy.
- ETC, INTA and IPGRI (ETC, Instituto Nacional de Tecnología Agropecuaria and the International Plant Genetic Resources Institute). 2002. Informe del Taller 29 y 30 de Julio del 2002 San Carlos de Bariloche, Argentina. Proyecto Pehuén. Conservación, Manejo y Uso Sustentable de los Recursos Genéticos de la *Araucaria araucana* en Argentina. Comunidades de Aucapán y Chiuquihuín. ETC Leusden, INTA San Carlos de Bariloche, IPGRI Rome, Italy.
- FAO (Food and Agriculture Organization of the United Nations). 1996. Report on the state of the world's plant genetic resources for food and agriculture. Prepared for the International Technical Conference on Plant Genetic Resources Leipzig, Germany, 17-23 June 1996. Rome, Italy.

- Friis-Hansen, E. and B. Sthapit (eds.). 2000. Participatory approaches to the conservation and use of plant genetic resources. International Plant Genetic Resources Institute (IPGRI), Rome, Italy.
- Ghimire, K.B. and M.P. Pimbert (eds.). 1997. Social change and conservation. Environmental politics and impacts of national parks and protected areas. Earthscan Publications, London, UK.
- Guha, R. 1997. The authoritarian biologist and the arrogance of anti-humanism: wildlife conservation in the third world. *The Ecologist* 27(1):14-20.
- Gunderson, L.H., C.S. Holling and S.S. Light (eds.). 1995. Barriers and Bridges to Renewal of Ecosystems and Institutions. Columbia University Press, New York, USA.
- Halwart, M. and G. Haylor. 2001. Participatory approaches for aquatic resources management and development. Thoughts and lessons collected by DFID (Department for International Development) and FAO during 2000. *In Utilizing Different Aquatic Resources for Livelihoods in Asia: A Resource Book*. International Institute of Rural Reconstruction (IIRC), International Development Research Centre (IDRC), Food and Agriculture Organization of the United Nations (FAO), Network of Aquaculture Centers in Asia-Pacific (NACA), and International Center for Living Aquatic Resources Management (ICLARM). Ottawa, Canada, Rome, Italy, and Manila, Philippines.
- Holling, C.S. 1986. Resilience of ecosystems, local surprise and global change. Pp. 292-317 *in Sustainable development of the biosphere* (W.C. Clark and R.E. Munn, eds.). Cambridge University Press, Cambridge, UK.
- Holling, C.S. 1995. What barriers? What bridges? Pp. 3-34 *in Barriers and Bridges to Renewal of Ecosystems and Institutions* (L.H. Gunderson, C.S. Holling and S.S. Light, eds.). Columbia University Press, New York, USA.
- IPGRI. 2003. Report of the final synthesis workshop of the project, Conservation, Management and Sustainable Use of Forest Genetic Resources with Reference to Brazil and Argentina, held in San Carlos de Bariloche, Argentina. International Plant Genetic Resources Institute (IPGRI), Rome, Italy.
- IPAP (International Programme Advisory Panel). 1997. IPAP discussion paper on community participation. International Planned Parenthood Federation (IPPF). <http://www.ippf.org/ippf/compart>.
- Jiggins J. and H. de Zeeuw. 1992. Participatory development in practice: Process and methods. *In Farming for the Future. An Introduction to LEISA* (C. Reijntjes, B. Haverkort and A. Waters-Bayer, eds.). Macmillan, London, UK.
- Leach, M., R. Mearns and I. Scoones. 1997. Community-based sustainable development: consensus or conflict? *Institute of Development Studies (IDS) Bulletin*, 28:1-3.
- Lee, K.N. 1993. Compass and gyroscope. Integrating science and politics for the environment. Island Press, Washington, DC, USA.
- Martin, A. and J. Sherington. 1997. Participatory research methods – Implementation, effectiveness and institutional context. *Agric. Sys.* 55(2):195-216.
- Merrill-Sands, D. 1986. The technology applications gap overcoming constraints to small farmer development. FAO Research and Technology Paper no. 1. Food and Agriculture Organization of the United Nations (FAO), Rome, Italy.
- Oltheten, T.M.P. 1995. Participatory approaches to planning for community forestry. Results and lessons from case studies conducted in Asia, Africa and Latin America. Food and Agriculture Organization of the United Nations (FAO), Rome, Italy.
- Pretty, J.N. 1994. Alternative systems of inquiry for a sustainable agriculture. *Institute of Development Studies (IDS) Bulletin*, 25(2).
- Pretty, J.N. 1995. Regenerating Agriculture. Policies and Practice for Sustainability and Self-reliance. Earthscan Publications, London, UK.
- Reij, C. and A. Waters-Bayer. 2001. Farmer innovation in Africa: a source of inspiration for agricultural development. Earthscan Publications, London, UK.
- Reijntjes, C., B. Haverkort and A. Waters-Bayer. 1992. Farming for the future. An introduction to LEISA. Macmillan, London, UK.

- Röling, N.G. and M.A.E. Wagemakers. 1998. *Facilitating Sustainable Agriculture. Participatory Learning and Adaptive Management in Times of Environmental Uncertainty*. Cambridge University Press, Cambridge, UK.
- Shannon, M.A. 2002. The use of participatory approaches, methods and techniques in the elaboration of integrated management plans. State University of New York School of Law, New York. <http://www.bova-university.org/html/edu/msccourses/forestry/Shannon%202002b.doc>
- Systemwide Program on Participatory Research and Gender Analysis. 1997. A global programme on participatory research and gender analysis for technology development and organizational innovation. AgREN Network Paper no 72. Overseas Development Institute, Agricultural Research and Extension Network (AgREN), London, UK.
- van Veldhuijzen, L., A. Waters-Bayer and H. de Zeeuw. 1997. *Developing technology with farmers. A trainer's guide for participatory learning*. ZED Books, London, UK.
- Waisbord, S. 2001. *Family tree of theories, methodologies and strategies in development communication. Convergences and differences*. Prepared for the Rockefeller Foundation. <http://www.comminit.com/pdf/familytree.pdf>

Chapter 13

The practical implications of research outputs from forest genetic studies

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1. Introduction

Translating forest genetic research results into practice is fundamentally necessary for conservation success, although it poses many challenges. These challenges are associated with issues such as scaling up outcomes from investigations of limited geographic scale, generalizing research results to species and settings other than those studied, and finding or creating sociopolitical environments favourable to adopting proposed solutions.

In this chapter, the practical implications of forest genetic studies are presented. We introduce a portfolio of frequently asked management questions posed by different forest-user groups for which forest genetic studies can offer answers. We also present examples of investigations that have addressed a range of management issues. Most of the examples presented are from case studies undertaken by the International Plant Genetic Resources Institute (IPGRI) in collaboration with other institutes or research centres in developing countries. Some of these examples highlight the difficulties faced when trying to bridge research findings with policy-making and practical actions.

2. How research results can be translated into practice

Concern for the world's forests has prompted the design and implementation of many initiatives to promote conservation of forest genetic resources (FGRs). But in order to achieve this goal, many practical considerations must be understood and addressed, such as the need for local group involvement in the conservation and sustainable use of species that are simultaneously local, regional and global resources, the need for managers to recognize when and under what circumstances species and populations are threatened, and the need to identify suitable solutions that are relatively cost-effective and easy to implement.

There are many (other) social, political and economic factors such as population growth, inequitable land and wealth distribution, and/or conflicting government policies that may pose even greater constraints to the implementation of conservation strategies for genetic resources than do technical or biological deficiencies (Ledig 1986), but these are not addressed in this chapter.

Here we present biological research questions associated with the various strategies for conservation (*in situ* and *ex situ*) and use of FGRs and which reflect the perspectives of many types of forest resource users: individuals or groups within local communities, forest managers, logging companies, policy-makers, conservation agencies, or nongovernmental organizations (NGOs) and other types of organizations addressing conservation or use of forest resources.

Conservation – in situ

- *What is the potential of economically important, priority species to adapt to climate change?* Loss of variation in adaptive traits will usually reduce the rate at which species can evolve in response to environmental changes, thereby increasing their risk of extinction. Forest genetic studies contribute to the understanding of adaptive mechanisms in tree species and support ‘active’ forms of gene management in preparation for response to rapid environmental and climate changes.
- *Which species are more sensitive to threats?* Vulnerability of species depends on their biological traits and the selection pressures to which they are exposed. Genetic diversity is particularly relevant to the long-term evolution of species. Genetic erosion vulnerability maps or indices are useful tools in helping to identify those species that require priority conservation action.
- *For more-or-less intact ecosystems containing priority species, where do we establish in situ conservation areas?* What is their optimal size? These questions, which are particularly important to policy-makers and natural resource conservation organizations, can be answered in part by undertaking a set of research activities aimed at characterizing how intraspecific diversity is structured and spatially distributed for target species, thereby helping to define the locations and the minimum sizes of areas to be preserved.
- *When natural ecosystems have become seriously fragmented, what size and shape should fragments have to be in order to maintain minimally adequate diversity in target species, and what is the degree of ecological connectivity that will allow adequate gene flow?* Species respond differently to forest fragmentation and isolation depending on their ecological and reproductive traits and their interactions with pollinators and seed dispersers, which are also affected by fragmentation. Genetic studies can determine the level of genetic erosion resulting directly or indirectly from reduced gene flow. They are also useful for assessing the genetic (gene flow) effectiveness of forest corridors.

Conservation – ex situ

- *What provenances are needed to recover target tree species?* The translocation of organisms during the restoration of native ecosystems has generated a myriad of questions on sampling protocols and intraspecific hybridization between locally adapted and transplanted genotypes. Studies are now in progress by IPGRI and other researchers to determine both the extent of local genetic adaptation and the potential risks of introducing foreign genotypes, including founder effects, genetic swamping and outbreeding depression. Furthermore, novel molecular markers are being used to predict the genetic consequences of translocation across a population; in combination, such markers have great power to determine appropriate seed transfer zones in restoration planning for native plant populations.
- *Where, how and when do we collect material for ex situ conservation?* In some cases, ex situ conservation may be the only option for safeguarding priority species during short periods of time, before augmentation or reintroduction can occur in natural environments or the species is used in planting programmes. Forest practitioners and managers need access to seed sources with known origins that are properly characterized from a genetic point of view. Furthermore, basic protocols for seed collecting, handling, storage, germination and bulking must be applied, and where they do not exist they must be developed.

Sustainable use

- *What are the sustainable levels of exploitation for wood and nonwood forest products (NWFPs) in target species?* Harvesting timber and extracting NWFPs from forests both contribute to the livelihoods of forest-dwelling communities. However, the use of forest resources affects the genetic diversity of many species. After clear-cutting, harvesting of reproductive structures from trees for food or medicine is likely to have the most significant impact on genetic diversity, especially when harvests are substantial,

extensive and unregulated. Research on the genetic processes of trees harvested for NWFPs can help achieve a balance between the needs of forest-dependent peoples and the genetic integrity of the resource.

- *How widespread and how damaging is genetic pollution in target species?* During the last decade, genetic pollution by pollen dispersal and/or hybridization has become a substantial concern for conservationists and forest managers. The risk of escape of transgenes from genetically modified organisms has focused public attention on the genetic pollution of native gene pools by introduced exotic species, nonlocal provenances and highly selected or genetically modified genotypes. More studies on genetic pollution are needed to provide a scientific basis to the regulatory frameworks that control the movement of genetic material within and across national boundaries. In the next sections, the research topics outlined above will be discussed in more detail. We use IPGRI case studies, particularly results from this project funded by the German Federal Ministry for Economic Cooperation and Development (BMZ), as background.

3. Examples of practical applications from genetic studies of FGRs

3.1 Conservation – *in situ*

Identification and conservation of adaptive genetic variation

Worldwide, the overall objective of the conservation of FGRs is the maintenance of genetic diversity in the thousands of tree species of known or potential socioeconomic and environmental importance. As the level and distribution of genetic variation in any species are expected to be in a process of constant natural change resulting from the main forces of evolution, the concern of conservation should be to preserve the evolutionary processes that promote and maintain genetic diversity, not on endeavours to preserve the present distribution of variation as an end in itself (Namkoong *et al.* 1997; Namkoong 2001).

Thus, *in situ* conservation is usually the preferred strategy for most wild plant species, including the wild relatives of domesticated crops, because its dynamic nature permits ongoing evolution. It not only allows populations of interest to continue to be exposed to evolutionary processes, it also normally implies that rapid genetic erosion and strong directional change in genetic composition are avoided.

Genes affecting the expression of adaptive traits fundamentally influence an individual organism's capacity to survive and reproduce, and the maintenance of adaptive trait variability in plant and animal species' gene pools is emerging as a primary objective of biodiversity conservation (Franklin 1980; Gregorius 1991; Eriksson *et al.* 1993). This is made especially important by the environmental and climatic changes that are now imposing substantial pressures on plant adaptive mechanisms, especially in forest tree species.

Adaptive trait variability has thus become a conservation and research priority in forest biodiversity management (Ennos *et al.* 1998). Nevertheless, it is hard to identify genome portions that contain genetic variation of adaptive significance (Smith and Wayne 1996; Lynch *et al.* 1999). And forest managers must also contend with genetic changes in pests and pathogens, changes that may intensify with growing climate change (Dale *et al.* 2001; Scherm 2004). Forest geneticists must therefore identify the genetic control mechanisms of resistance, while also ensuring that adequate genetic variation is maintained in breeding populations in order to provide this resistance.

Adaptive traits are usually quantitative, controlled by multigenic determiners, and their expression and variability have been affected by natural selection through environmental interactions (Falconer and Mackay 1996). However, neither geneticists nor forest managers can monitor and manage all loci relevant for conservation for three main reasons (Sherwin and Moritz 2000): (i) it is unlikely that the same set of loci is consistently

responsible for the adaptive traits of interest under all conditions; (ii) although associations of gene diversity and fitness are often significant and positive, the range of values for the fitness component itself is often narrow, so it is unlikely that a particular locus will have an identifiable effect on fitness; and (iii) in short-term experiments, it is difficult to distinguish between the effects of heterozygosity at individual loci and heterozygosity at linked loci on the same chromosomal segment. Estimating adaptive traits for even one species is hard because it necessitates the observation of a large number of populations for adaptive traits in a range of environments. This is expensive and time-consuming, and impractical for many tree species.

Thus, what can we monitor and manage within single populations in order to conserve adaptive genetic variation (whether multi or single locus)? It is usual to use molecular variation observed through genetic approaches as a surrogate for adaptive variation (Lynch *et al.* 1999). Another frequently used surrogate is the maintenance of large population size, based on a generally positive correlation between population size and genetic diversity (Frankham 1996). But this method should be used cautiously because of the exceptions to the correlation mentioned above (Leberg 1993), the difficulty of calculating effective population size from census and demographic data (Luikart and England 1999), and problems related to the determination of effective population size (Franklin 1980; Lande 1995). In relation to the spatial distribution of intraspecific genetic diversity, existing theory suggests that at least some components of a species' adaptive diversity can be retained by maintaining viable populations across the range of environments it occupies, called evolutionarily significant units (ESUs), and preferably within connected and heterogeneous landscapes. And Lesica and Allendorf (1995) conclude that the conservation of peripheral populations is particularly important for the maintenance of evolutionary potential.

Pioneering studies have attempted to compare molecular genetic diversity with adaptive variation at within-species levels (Kremer 1994; Lynch 1995). As scarce as they are, data on adaptive traits are available for some economically important species where populations have been observed using rigorous experimental designs and across many environments (e.g., Pederick 1979; Alia *et al.* 1997).

In studying adaptation to climate change, the spatial distribution of biological diversity and climatic data have been used to generate models to predict the impact of climate on the geographic distribution and genetic diversity of some tropical forest tree species (Enquist 2002). Studies of past responses of species to disturbance have also helped in understanding the effects of current environmental changes and in designing conservation strategies (McLachlan and Clark 2004).

Work like this has occurred at IPGRI in collaboration with the International Centre for Tropical Agriculture (CIAT). Information on the spatial distribution of genetic diversity was coupled with climate models using the geographic information system (GIS)-based modelling tool FloraMap. This enabled researchers to predict shifts in the ranges of 18 important crop species and wild groundnuts (*Arachis* spp.) in South America (Hijmans *et al.* 2001), as well as the extinction of distinct populations caused by habitat loss as a consequence of climate change. The scenarios painted by these authors point to a precarious future for most of the 18 species analyzed. Nevertheless, efforts are in progress to develop conservation strategies for several wild *Arachis* species in Bolivia, said to be the origin of the domesticated *Arachis* plant (Stalker and Simpson 1995).

Such predictive tools now allow researchers to identify endangered, fragmented or marginal populations of important forest trees species, thereby guiding the collection of material to be preserved in gene banks and, ultimately, to be used in augmentation or recovery programmes.

Priority setting in tree species conservation

Given the urgent situation and the limited resources available, conservationists must choose where to focus their attention. Moreover, biodiversity is not distributed evenly and some biotas and ecosystems are more complex, distinctive or threatened than others.

For these reasons, identifying priority areas and/or species and associated conservation activities is critical for developing effective conservation strategies.

Studies of past and ongoing extinctions frequently describe nonrandom patterns of species losses across taxa (Raup 1994; McKinney 1997; Russell *et al.* 1998; Purvis *et al.* 2000), while studies of extant populations and species have identified factors associated with vulnerability to extinction (Laurance 1991; Gaston 1994; Woodroffe and Ginsberg 1998; Foufopoulos and Ives 1999). It is well known that the main processes driving extinction are the 'evil quartet' (habitat loss, over-exploitation, introduced species and chains of extinction; Diamond 1984). But statistical testing for differences among species in terms of extinction vulnerability has proven to be difficult. Problems encountered include incomplete information on species' biology, conservation status and phylogeny, and on the interrelationships between possible predictor variables (McKinney 1997; Woodroffe and Ginsberg 1998).

Habitat loss inevitably leads to a reduction in population size; it has been known for more than 150 years that small populations of plants and animals face a higher risk of extinction than larger ones (Darwin 1855). Genetic factors may sometimes contribute to the failure of small populations to survive – whether it is from a lack of appropriate genetic diversity or an accumulation of deleterious mutations that lowers average reproductive rates in populations (Barret and Kohn 1991; Sherwin and Moritz 2000) – although a lack of genetic diversity in small populations is more likely to be a symptom of endangerment than its cause (Holsinger and Vitt 1997).

However, since the work of Lande (1988) there has been a rethinking of the role of genetic data and processes in the management of rare species. For example, many conservationists now ask such questions as 'How expensive is it to obtain the genetic data?' and 'What is the relationship between the genetic structure of a population and its demographic processes?' In response, we would say the advantage of a model that accounts for genetic variation is that it forces one to be explicit about the effects of inbreeding, outbreeding depression, mutational meltdown and a range of plausible interactions between demographic attributes and genetic composition (Frankham 1995).

Setting species conservation priorities based on the level of threat they face and their importance is a common exercise that should precede conservation action, and many criteria have been adopted by different organizations to help them set priorities in terms of targets, approaches and scales of intervention (Redford *et al.* 2003). To guide priority setting in conservation actions, categories have been defined that identify different levels of threats and vulnerability; lists of species have been compiled (e.g., The World Conservation Union [IUCN], <http://www.redlist.org>); and keystone species have been identified (i.e., species that have the capacity to significantly influence ecosystem stability). Furthermore, Henle *et al.* (2004) have conducted studies to identify predictors of tree species' sensitivity to threats, such as fragmentation, based on a set of demographic, life-history, community and biogeographic traits. Predictors also help understand species' proneness to extinction and therefore can aid in identifying those species most in need of conservation.

IPGRI has been deeply involved in wide regional consultations aimed at identifying priority species for trans-boundary FGR conservation. In fact, at the 13th session of the Food and Agriculture Organization (FAO) of the United Nations Committee on Forestry (COFO), and the tenth and subsequent sessions of the FAO Panel of Experts on Forest Gene Resources, it was recommended that FAO, in collaboration with partners like IPGRI, assist countries to convene regional and sub-regional workshops to support the development of action plans for the conservation and sustainable use of FGRs. One of the highlights of the regional workshops held thus far has been the need to develop common approaches to priority setting, and especially for species that would benefit from genetic management. It was also recommended that mechanisms be in place to ensure that needs and aspirations of local communities are duly considered (Quédrago and Boffa 1999; Palmberg-Lerche and Hald 1999). An example of IPGRI's work on the selection of priority tree species for research and conservation is presented in Box 1 (overleaf).

Box 1. Priority setting for FGR conservation in Africa

Forests have incalculable value for the environment as well as for food, medicine, energy and other human needs. Their preservation is a global challenge. In 1998, IPGRI initiated the Sub-Saharan Africa Forest Genetic Resources Programme (SAFORGEN) to ensure that FGRs in this part of the world received the conservation and research attention they required.

Countries within SAFORGEN set network priorities in terms of species and actions; determine national and collaborative activities; and disseminate results of activities to network members. Thus far, member countries have proposed to target priority species within four pilot sub-networks: food tree species, wood and fibre tree species, medicinal trees species and fodder tree species. Sponsored by the United Nations Environment Programme (UNEP) and IPGRI, field studies have been undertaken on the genetic diversity and the impact of human practices on two fodder tree species in Benin, two food species in Kenya and two medicinal tree species in Togo. Regional training initiatives on the conservation and sustainable use of FGRs have been organized through the network.

Where should genetic conservation areas be established?

The status of forest biodiversity is determined both at the level of the community (such as ecosystems and habitats) and at various levels of its constituent parts (e.g., species populations or genomes). However, forest conservation has often been indirect and one dimensional, resulting from the establishment of national parks, as these have usually been created for the protection of megafauna, spectacular scenery or rare ecosystems, with little attention paid to the hierarchy of biological organization or the spatial and temporal dynamics of biological processes. Furthermore, the criteria for selecting conservation areas differ according to the conservation objectives, and areas selected to conserve ecosystems, species or genes may not coincide with each other. While awareness that there is a genetic aspect to forest conservation is growing, the level of understanding of the principles behind the issues and of the genetic implications for conservation and management are limited.

Some scholars see genetic research as a way of monitoring the 'information content' of biodiversity (Crozier 1997). Crozier's thesis is that a utilitarian basis for preserving tree species, as opposed to a moral or aesthetical basis, depends upon their information content, most notably on the identification of genes that might be important to humans. Genetics can be used to assess this 'information content' in tree species and thus, by extension, can also help identify the location of biodiversity hotspots. An objective basis for comparing the information content of sets of taxa lies in various measures that evaluate their 'phylogenetic diversity'. Petit *et al.* (1997) note, however, that conservation priorities need to consider uniqueness as well as diversity. These authors suggest that a measure of 'allelic richness', analyzed by rarefaction techniques, provides an unbiased estimate of diversity and uniqueness across different sample sizes and tree taxa.

Another of the applications of genetics to the selection of reserve locations lies in delineating biogeographic zones that are centres of endemism. Studies of the phylogeny and population genetic structure of a few well-chosen genera and families may point to patterns of differentiation that can be extrapolated to protect many species. Attempts have thus been made to maximize the capture of phylogenetic diversity in the selection of conservation areas (Rodrigues and Gaston 2002), and to determine whether genetic diversity or/and taxonomic richness can be surrogates for phylogenetic diversity (Polansky *et al.* 2001). While there is a positive relationship between phylogenetic and genetic diversity, these studies showed that using only taxonomic richness as a criterion for reserve selection may introduce biases towards sites containing many closely related species,

while selection based on phylogenetic diversity will tend to select sites with monophyletic taxa. Unfortunately, data on the phylogenetic relationships among tree species can be scarce and/or incomplete, and this can hinder their widespread application to reserve planning (Polansky *et al.* 2001).

Box 2 introduces IPGRI's phylogenetic studies on neotropical forest tree species in collaboration with the Smithsonian Tropical Research Institute.

Analyses of the spatial distribution of forest genetic diversity are important to the decisions that are made on *in situ* conservation locations, as the ultimate goal of *in situ* forest conservation is to maintain broad genetic diversity so that tree species retain the potential to adapt to environmental change. This means that dynamic approaches to conserving genetic diversity are generally much more useful in this regard than approaches that focus on the static conservation of individual genotypes. A successful *in situ* gene conservation programme must thus fulfil three basic requirements (Koski *et al.* 1997): (i) regeneration of target tree populations must be assured with the new generation predominantly resulting from mating within the conserved populations; (ii) the number of genotypes in the conserved populations must be large enough to include most of the common alleles; and (iii) the conserved stands must be distributed in such a way as to cover the spatial genetic variation present in the species. For each site where a species is found, genetic, demographic and ecological factors must be considered in devising the most appropriate *in situ* conservation strategy.

Molecular genetics and other biotechnology tools are being used more and more to assess levels of genetic diversity in tree species (Boshier 2000), and this has implications for determining forest conservation locations.

An example of research undertaken to identify the most suitable *in situ* forest conservation areas in Europe is the framework of the European Forest Genetic Resources Programme, EUFORGEN (<http://www.euforgen.org>). This is a collaborative mechanism among European countries to promote conservation and sustainable use of FGRs, coordinated by IPGRI in technical collaboration with the FAO. Figure 1 (overleaf) presents a flow chart with an overview of the conditions for the selection of the most appropriate *in situ* conservation strategy for the European black poplar (*Populus nigra*). The overview is based on work carried out by the EUFORGEN *Populus nigra* Network.

Box 2. Phylogenetic studies on a neotropical forest tree species

Biogeographic history contributes to the pattern of tree species' diversity across forest ecosystems by exerting influences on regional differences in speciation, extinction and immigration (Ricklefs and Schluter 1993). Historical contribution can be evaluated with paleontological or phylogenetic approaches. This second approach was adopted by researchers at the Smithsonian Tropical Research Institute to reveal differentiation in a widespread tropical tree, *Symphonia globulifera* L. f. (Clusiaceae), during the Tertiary period (Dick *et al.* 2003). In order to initiate historical analyses of such widespread neotropical trees, the nuclear ribosomal internal transcribed spacer (ITS) region was sequenced in populations of the species that occur in the neotropics and West Africa. The development and application of an ITS molecular clock also enabled researchers to estimate the time at which the populations diverged. These findings supported evidence from paleontological data that indicated three marine dispersal events of this species leading to the colonization of Mesoamerica, the Amazon basin and the West Indies. A strong phylogeographic structure was found among trans-Andean populations of *S. globulifera*, while ITS nucleotide variation was absent across the Amazon basin, suggesting marked regional differences in the demographic history of this rainforest tree species.

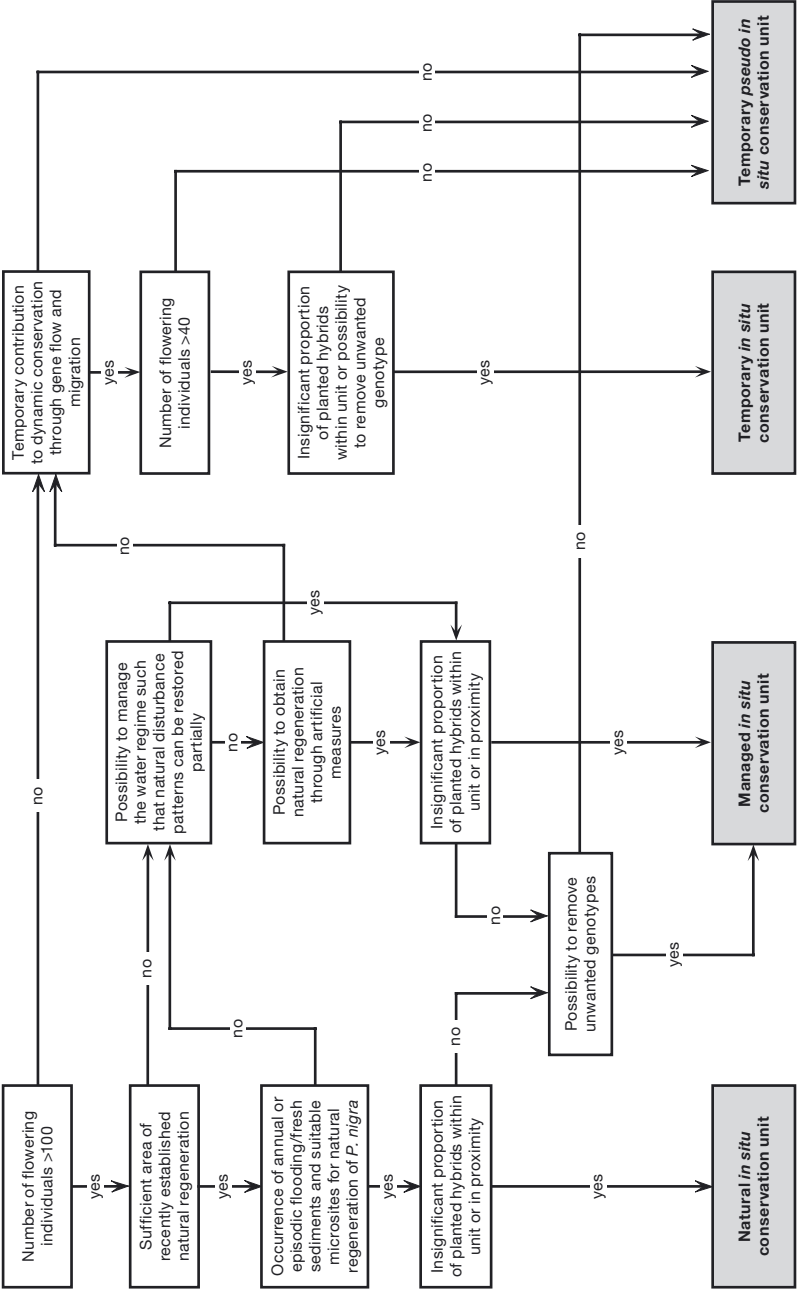


Figure 1. Decision-making flow chart for selecting the most appropriate *in situ* conservation strategies for *Populus nigra* (from Lefèvre et al. 2001).

Assessing the impact of forest fragmentation on the intraspecific diversity of important tree species

Forest fragmentation resulting from the conversion of forests to other land uses is an increasing reality in most tropical countries and now represents a critical area of forest genetic research. Genetic variation in most tropical tree species has been known to be high for some time, with most variation residing within rather than among populations, indicating extensive gene flow within species (Hamrick and Loveless 1989). A general concern among forest managers and researchers today is that forest fragmentation is leading to reduced gene flow and increased genetic drift in forest tree species, potentially resulting in permanent loss of genetic diversity (Gilpin and Soulé 1986). However, fragmentation has also been found to lead to increased levels of gene flow among some fragmented populations and, hence, the maintenance or even increase of genetic diversity within these populations (Hamrick 1992; Young *et al.* 1996). More genetic research is thus needed on fragmentation and isolation responses in different groups of forest tree species so that meaningful generalizations can be drawn when conservation guidelines are being developed.

It is also known that habitat fragmentation can indirectly affect forest genetic diversity by disrupting mutualisms such as pollination and dispersal (Levin and Kerster 1974; Aizen and Feinsinger 1994). This can lead to both genetic isolation of populations and reductions in the population size and genetic diversity (Young *et al.* 1996), thereby adding less obvious negative genetic effects to the direct effects of habitat loss. Formulating forest management strategies to conserve tree species requires an understanding of the consequences of fragmentation on the level and spatial distribution of genetic diversity and on the extent of connectivity among stands for gene flow.

Our understanding of the impact of human activities on mating systems, gene flow and genetic diversity in trees also relies on the continued discovery and development of genetic markers that permit researchers to estimate gene flow and effective population size in trees. The development of highly variable microsatellite markers, and increased recognition of their importance in determining genetic consequences of fragmentation, have led to more research in this area (Chase *et al.* 1996; Dawson *et al.* 1997; Aldrich and Hamrick 1998; Nason *et al.* 1998; Dayanandan *et al.* 1999).

Collaborative research between IPGRI and the universities of Costa Rica and Massachusetts (United States of America) is addressing several of these issues, including the characterization of new genetic markers for trees and the impact of forest fragmentation on the reproduction and gene flow of tropical dry forest species used for timber in Costa Rica (Box 3, overleaf).

3.2 Conservation – *ex situ*

Sampling the distribution of forest genetic diversity for *ex situ* conservation

Forest habitat destruction is happening at alarming rates all over the planet. Little is known of the patterns of threats and genetic erosion at ecosystem, species and intraspecific levels, but the survival of thousands of tree species will in all likelihood ultimately depend on their conservation outside of natural areas, for example, in managed forests, agricultural landscapes, botanical gardens and arboreta, and seed banks or field gene banks. Huge gaps exist in our knowledge of the distribution of genetic variation in forest trees. Through the collection and storage of standard seeds via reduced temperature and humidity techniques, cryogenic preservation of tissue and recalcitrant seeds, and controlled temperature tissue culture, *ex situ* conservation has proven itself to be a flexible complement to *in situ* conservation. It is also an efficient way of capturing the rare alleles that represent the most vulnerable component of diversity.

In order for *ex situ* conservation to obtain minimally adequate levels of genetic diversity in trees, carefully conceived sampling strategies are needed. Thus, before implementing *ex situ* conservation strategies, genetic resources must be selected based on knowledge

Box 3. Impact of fragmentation on FGRs in Costa Rican forests

The impact of fragmentation on the reproductive biology of *Enterolobium cyclocarpum* in Guanacaste National Park in Costa Rica was the subject of a research project coordinated by IPGRI. The study compared the reproductive characteristics of *E. cyclocarpum* trees growing in continuous forests and in pastures. Researchers hypothesized that a reduction of continuous habitat would be accompanied by lower rates of flower visitation by pollinators and thus by the likelihood of a decline in pollination, fruit set and seed production per fruit. Nevertheless, in fragmented forests, the number of flowers with pollen on their stigma and the number of pollen grains deposited were found to be similar to those in continuous forest. However, in the fragmented forests, only a limited fraction of trees with pollinated flowers later produced mature fruits. This suggested that factors other than pollination were causing the reduced fruit maturation. Since plants may regulate the quality of their progeny, researchers then hypothesized that competition among developing fruits and/or genotypic interactions between paternal and maternal parents were responsible for the differential seed abortion. Because progeny vigour in seedlings from pasture trees was significantly lower than in seedlings from continuous forest, it was assumed that habitat fragmentation either disrupted mechanisms regulating progeny vigour and quality, or that progeny vigour and quality were affected by inbreeding. But the research also revealed that fragmentation had no effect on outcrossing rates. Furthermore, the study demonstrated how important isolated trees in fragmented landscapes are for the movement of pollinators and for gene flow while, on the other hand, also showing that seeds from fragmented sources are inferior to continuous forest sources for the establishment of commercial plantations (Rocha and Aguilar 2001).

Another study was undertaken in Guanacaste National Park to identify and characterize simple sequence repeat (SSR) loci in *Carapa guianensis* that could then be used to detect polymorphisms in related species. Employing these markers, the study tested the effects of fragmentation on gene flow and genetic differentiation among fragmented and continuous forest populations, comparing individuals from pre-fragmentation adult cohorts with individuals from post-fragmentation sapling cohorts. Genetic distances between adults and saplings were greater in forest fragment populations than in continuous forest. For the site subjected to cattle grazing and selective logging, average allelic richness of the sapling cohorts was lower than in the adult population, while the opposite was true for the ungrazed and selectively logged continuous forest populations. Moreover, the larger genetic distances among all cohorts when compared to adult cohorts was mostly attributable to juveniles and, more specifically, to the substantial differentiation that existed between juveniles in the fragmented populations and those from the continuous populations. The project generally indicated that fragmentation decreases gene flow and increases allelic differentiation among populations.

of spatial patterns of variation. Once these various parts of the conservation process are known and acted upon, the final step of the conservation programme is to regenerate the resource (Finkeldey and Hattermer 1993; Hattermer 1995).

The literature treating the theory and rationale of optimal sampling strategies for *ex situ* conservation is very rich (e.g., Allard 1970; Marshall and Brown 1975; Namkoong 1988; Crossa 1989; Krusche and Geburek 1991; Lawrence *et al.* 1995).

The provenance of the seed is generally recognized as the key unit of the genetic resource when collecting occurs (Matyas 1996; Brown and Hardner 2000). Many studies have highlighted the existence of genetic divergence among provenances of the same species (e.g., Stern and Roche 1974, Mikola 1982). Brown and Hardner (2000) state that the region or

area where the tree species occurs should be divided into distinct ecogeographical zones, and they provide guidance on the number of populations – and the number of individuals in each population – from which seeds should be randomly collected.

Microsatellites have been used in tree species to investigate the spatial distribution of genetic diversity across landscapes in order to identify areas most suitable for *ex situ* conservation collecting. Modelling tools are being developed to determine the most efficient sampling strategies once collecting sites are identified.

The physical and financial limitations of most *ex situ* conservation methods will determine what type and amount of genetic variation can be sampled and therefore conserved. Although it is always desirable to capture as much genetic variation as possible, the total number of trees sampled must be manageable.

IPGRI's FGR programme is investigating the amount and spatial organization of genetic variation in tree species with contrasting modes of distribution, dispersal, pollination and seed dispersal. For rare and endangered species, genetic studies are integrated with demographic and ecological research to develop management and restoration plans.

An example of this work is the research undertaken by IPGRI in the Middle East, mainland southeast Asia and sub-Saharan Africa, designed to locate genetic diversity in timber and nonwood forest species of economic importance or in species threatened with extinction (Box 4, overleaf). This research has identified distributional ranges of selected species, clarified their taxonomy and investigated their pollination and seed dispersal behaviours.

IPGRI has also worked on *ex situ* storage problems for forest trees intolerant of desiccation (Box 5, on p. 257).

Selection of suitable provenances for restoration and rehabilitation purposes

Forest restoration for conservation purposes is defined by Brown and Lugo (1994) as the deliberate alteration of ecological patterns and processes for the purpose of recreating some presumed set of natural, pre-disturbance ecosystemic conditions. Restored forests should be similar in structure, function and composition to the historic forests of the region prior to disturbance by humans (Lamb *et al.* 1997). Some ecological factors relevant to the establishment of original forest conditions at various temporal and spatial scales are: (i) use of indigenous species; (ii) incorporation of natural successional dynamics; (iii) consideration of ecological relationships; and (iv) consideration of the effects of ecological patterns of species on ecosystem processes.

The degree of isolation of a restoration site in relation to a seed source is a major factor limiting natural dispersal to the site. Ideally, restoration interventions should occur in areas neighbouring native forest seed sources. If this is not possible, consideration should be given to the development of planted forest corridors through which forest-dwelling seed dispersers may travel to the restoration site. There are also examples of forest plantations established on degraded sites long devoid of native tree cover that have acted as successional catalysts, that is, they have facilitated the recolonization of native flora through their influence on understorey microclimate and soil fertility, suppression of dominant grasses, and provision of habitats for seed dispersing animals (Parrotta 1995). Properly selected provenances, well adapted to local environmental conditions, should be identified for the purposes just mentioned. Careful determination of genetic variation and the distribution of FGRs should be carried out before initiating restoration interventions.

The proper selection of sites from where regeneration material should be collected is a fundamental part of the forest restoration process. This is because when seeds are planted away from their native environment, saplings and trees may suffer from maladaptation, leading to pest attacks, slow production and/or poor form. Seedlot registration systems are a simple way to minimize maladaptation by classifying administrative regions into ecologically similar 'seed zones', and they discourage the transfer of seedlots across zones (Conkle 1997). Reforestation projects frequently suffer reduced genetic diversity that could compromise their stability and resilience because they lack guidelines that

Box 4. Ecogeographic, socioeconomic and genetic surveys of tree species in central and west Asia and north Africa (CWANA)

In collaboration with local research partners (Lebanon: American University of Beirut, Saint Joseph University–Beirut, and Directorate of Rural Development and Natural Resources of the Ministry of Agriculture; and Syria: University of Tishreen–Lattakia, Arab Centre for the Studies of Arid Zones and Dry Lands, and Ministry of Agriculture and Agrarian Reform), IPGRI undertook ecogeographic surveys of several tree species important to the local economies of the CWANA region. Country ranges or when possible natural ranges of species were surveyed, with subsequent infraspecific genetic diversity investigated using genetic markers. Sampling strategies for these latter studies were based on results from the ecogeographic surveys. The species studied and the activities undertaken are described below.

In Syria, exhaustive ecogeographic, socioeconomic and genetic diversity surveys were undertaken on pistachio (*Pistacia vera*), an important arboricultural crop in the CWANA region. Because many of its wild relatives are sources of genes that confer pest and disease resistance in cultivated pistachios, several natural areas that include wild pistachios have been conserved, especially in areas with poor soils or that are subject to drought. Surveys produced an understanding of the diversity of cultivated varieties of pistachio in Syria and also of one of its common wild relatives, *Pistacia atlantica*. The project produced distribution maps of pistachio genetic diversity in the country and an assessment of the threats to the two species. The agrobotanical characteristics of *Pistacia vera* were also determined.

In central Asia, IPGRI is supporting the national FGR programmes of Kazakhstan, Kyrgyzstan, Tajikistan, Turkmenistan and Uzbekistan as these countries survey, collect, study and rescue remnant wild populations of pistachio. Until now, these trees had mostly been used as sources of nuts and firewood by local people. Wild pistachio populations will now also be used to develop new and better yielding varieties for the rural areas of central Asia.

The distribution of carob (*Ceratonia siliqua* L.) populations and their genetic variability in Syria and Lebanon were investigated. Carob is an important multipurpose species native to the Mediterranean basin. Precise data on carob's distribution and genetic diversity in Lebanon and Syria were not available, making it difficult to assess its conservation status. Carob trees have been selected and cultivated for centuries for the high pulp content of their pods, which are used for animal and human foods. Today, carob is receiving greater interest from industry, mainly from cosmetic and pharmaceutical companies. The tree is also valuable for its resistance to fire, which makes it attractive for use in vegetation corridors in fire-prone Mediterranean maquis. Because it commonly occurs in semi-natural areas near orchards in Lebanon, it was difficult to distinguish between cultivated and wild carob. In contrast, virtually every carob population in Syria seemed to be natural. But in this country, all the sites investigated were highly fragmented and threatened with habitat conversion. Leaf samples of the surveyed trees were analyzed using molecular markers to assess genetic diversity between and within populations in both countries.

specify minimally adequate numbers of trees and provenances from which seeds should be collected (O'Neill *et al.* 2001).

In contrast, when rehabilitating forests for commercial forestry purposes, either using native tree species and/or exotic species, the management goal is not to recreate the original (often more complex) forest ecosystem. The aim of such forest rehabilitation is to manage and utilize the rehabilitated forests to fulfil various human needs (Lamb 1994).

Rehabilitation studies for degraded forests, like for restoration research, should also try to understand the key factors that determine growth rates in small-scale plantations,

Box 5. Analyzing seed behaviour in forest tree species

The use of many high-value tropical forest species in tree planting and conservation programmes is hindered by seed handling and storage problems. Indeed, knowledge about seed physiology of most tree species ranges from scarce to nonexistent. Seeds from many tree species, particularly from the humid tropics, are difficult to handle because of their sensitivity to dry conditions and other physiological issues. Many are said to be recalcitrant or intermediate, meaning they are difficult to store even for short periods of time.

One project, initiated in 1996 and recently completed, screened 61 such tree species in 18 countries. It was coordinated by IPGRI and the Danish International Development Agency (DANIDA) Forest Seed Centre (DFSC) with funding from DANIDA. Results provided insights into the storage biology of economically important tree species, as well as into techniques that now give foresters and nurseries access to far greater numbers of tree species. Moreover, many species produced seeds that were not recalcitrant as was believed. Seeds from almost half the species examined could be stored for some time under reasonable controlled humidity and temperature conditions (<http://www.dfsc.dk/IPGRIproject.htm>).

develop guidelines for selecting species and acquiring genetic information from them, and improve technologies for site management. All of these will raise production and increase the chances for sustainability of plantations on degraded and low-potential sites.

Data are similarly needed to delineate 'seed transfer zones'. These are regions within which trees can be moved with little or no consequence to population fitness (Hufford and Mazer 2003).

While forest restoration and rehabilitation include a range of objectives, conceptual frameworks and techniques, both require sound management of FGRs for the long-term sustainability of the resource. Whether their objectives are restoration or rehabilitation, national systems of tree seed registration should adopt minimum standards for seed collecting that specify: (i) the number of parent trees required for each seedlot as well as the seed bulking methods (Marshall and Brown 1975); (ii) the genetic distance between parent trees that will minimize relatedness (Dawson and Were 1997); (iii) the yield and physiological qualities of the harvested seeds; (iv) the health of the mother trees; and (v) location data. Strict standards for documenting collecting localities will ensure that collectors can return to populations that perform well in particular environments for particular purposes, while populations that perform poorly can be avoided.

3.3 Sustainable use

How much can we use tree species for timber and nonwood forest products?

Commercial harvesting of timber by logging is the major form of forest disturbance in most countries. However, other forms of disturbance, such as NWFP extraction, fire or the cutting of wood for local consumption, can also threaten tree species, though the threat level from these disturbances will vary depending on the species' 'life history strategies' and mating systems (Noss 1983). Because natural forest ecosystems are composed of long-lived indigenous trees, tree populations can be used as indicators of anthropogenic stress. Genetic variability within tree populations is potentially a high indicator of forest ecosystem stability and permanence because it is a major factor in determining the adaptive potential of tree populations and their survival abilities under various stress scenarios (Kim and Hattener 1994; Baradat *et al.* 1995). Substantial research effort has thus been expended on the identification of criteria and indicators (C&I) for sustainable forest management, much of this within the framework of forest certification. C&I are neutral assessment tools for

monitoring forest trends that provide the means of measuring, assessing, monitoring and demonstrating progress towards achieving forest sustainability over time. C&I initiatives for sustainable forest management have been reviewed (Castañeda 2000; Castañeda *et al.* 2001; McKinnell 2002), and the Centre for International Forestry Research (CIFOR) is actively working on forest C&Is in the context of its Criteria & Indicators Toolbox series.

During the last decade, independent certification has gained momentum as an effective mechanism for improving and promoting sustainable management of forests (Pierce and Ervin 1999). A common element of certification bodies or programmes is that they operate within frameworks of clearly specified principles, criteria and indicators. Because C&Is are essential for defining or recognizing 'sustainable forestry' (Johnson and Cabarle 1993; Funston 1995), they offer tangible forestry management solutions by outlining conditions that must be met if forest management is to be deemed sustainable, rather than trying somehow to quantify the acceptability of forestry management (Boyle 2000).

A commonly heard complaint is that knowledge about the genetics and ecology of most tree species in tropical forests is still largely incomplete, and attempts have been made to include genetic and ecological information within C&Is. Genetic and ecological C&Is considered thus far have emphasized methods that characterize or visualize genetic variation and ecological processes, methods that detect temporal changes and trends in genetic and ecological parameters, the choice of thresholds or critical values to apply, and how to combine information from multiple indicators (Boyle 2000).

IPGRI's research on NWFPs currently focuses on understanding how patterns of NWFP use affect FGRs, the objective being to include thresholds for sustainable extraction of NWFPs into guidelines for forest management. A good example is the work on bamboo and rattan in Asia, the Pacific and Oceania (Rao and Ramanatha Rao 1999a and b). IPGRI is also coordinating research activities on C&Is in collaboration with IMAZON (Instituto do Homem e Meio Ambiente da Amazônia) and IMAFLORA (Instituto de Manejo e Certificação Florestal e Agrícola; Box 6) as well as participating in the activities of the Steering Committee of the Dendrogene project funded by DFID (the UK's Department for International Development) and undertaken by EMBRAPA (Empresa Brasileira de Pesquisa Agropecuária) in Brazil.

Limitations and challenges in developing and applying C&Is

Limitations in applying C&Is to forest management reside in the sometimes poor quality of the data collected and in the inevitable restriction of focus to a small number of species. And an important question remains on how to incorporate specific C&Is for assessing genetic diversity into overall assessments of management practices.

Challenges include creating C&I management guidelines that are clear, practical and easy to use, and that are based as far as possible on good science (ITTO 1998). Moreover, because costs (both financial and temporal) must always be contained, and because genetic studies can require sophisticated and expensive laboratory procedures (Boyle 2000; Jennings *et al.* 2001), another challenge is determining minimally adequate genetic indicators in the most efficient way possible. For example, should these indicators be based on direct measurements using molecular research or is there sufficient information available to allow genetic indices to be replaced by demographic indices such as population size, physical isolation, seed dispersal or number of mature individuals? Unfortunately, ecological processes occurring at large temporal and spatial scales are also difficult to monitor, and such information may not be readily available.

Unlike some economic and ecological C&Is, it is usually more difficult to 'translate' genetic indices into a practical and easily used form. Nevertheless, some scientifically sound genetic C&Is have been proposed by Stork *et al.* (1997), Boyle (2000), Brown *et al.* (2000) and Namkoong *et al.* (2002), but they are complicated and difficult to apply in practice (Jennings *et al.* 2001).

Tracking timber using genetic fingerprinting

In order to trace illegally logged timber, improved genetic fingerprinting methods have recently been developed that can identify commercial timber and its geographic origin.

Box 6. C&Is for sustainable forest management (SFM) in the Amazon basin

Fundamental information on the biology of forest tree species in complex tropical forest ecosystems is needed to create SFM plans, but it is often lacking, not properly organized into information systems, or hard to translate into forest management practices. While data already exist for some of the more important forest species, these are not always easily accessible (Martini *et al.* 1994; Hammond *et al.* 1996; Guariguata and Pinard 1998; Pinard *et al.* 1999; Sheil 1999; Jennings *et al.* 2001).

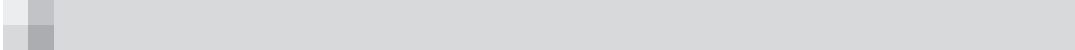
Research carried out from 2002 to 2004 by two major Brazilian NGOs (IMAFLORA and IMAZON) in collaboration with IPGRI focused on the generation of a database capable of supporting forest management decision-making. The database was designed to include genetic C&Is for sustainable use of forest resources in the Amazon. Project objectives were (i) to overcome dispersed data issues by collecting and organizing existing information on Amazonian forest tree species; (ii) to identify further research needs and especially to avoid duplication of efforts; and (iii) to create a pool of information on tree species that could be easily accessed by researchers, government agencies, timber companies, students, certification organizations and other users. The database was designed to assist the development of practical guidelines for both commercial forest management and conservation.

In order to assemble data on regional distributions of important timber species, information from the RadamBrazil project was used. Conducted in the 1970s, this is a large inventory of the forest resources found in the Amazonian region of the Transamazon Highway. A geo-referenced map of the RadamBrazil sampled sites was prepared and linked to the species data. The final database was spatially explicit, allowing for maps of tree species' distributions and densities to be produced across the Amazon and linked with other geo-referenced maps of soil, rainfall, forest type, topography and temperature.

Five timber companies also provided data on 40 commercial tree species. Within the framework of certification schemes, logging companies are required to do preharvest inventories. Thus, full forest inventories have been done for an estimated 70 000 to 100 000 hectares of the Amazonian region. This work was possible thanks to the collaboration of the certified logging companies Cikel Brasil Verde, Precious Woods Amazon e Pará, Gethal S.A. and Madeireira Juruá as well as the governmental and nongovernmental organizations Embrapa Amazônia oriental/ Projeto Dendrogene, Instituto Nacional Pesquisa da Amazônia (INPA), Instituto Brasileiro de Geografia e Estatística (IBGE), Superintendência de Desenvolvimento da Amazônia (SUDAM), Universidade Federal Rural da Amazônia (UFRA), and the Overbrook Foundation.

The database is still under development but will provide the most complete picture of timber species populations and management potential available for the Brazilian Amazon. Based on the information and tools to be generated by this project, recommendations and research priorities will be devised to improve forest management and conservation of timber species in this region.

These new methods are important technical advances for the forestry industry, for phytosanitary certification procedures and for forest tree conservation because they permit the chain of custody to be followed, thereby potentially ensuring the certification of timber from sustainably managed forests. Extensive genetic typing data sets now exist for several nontropical forest tree species. For the European white oak, a detailed reference map of chloroplast DNA variation across the species' range can now be used to track oak wood (Deguilloux *et al.* 2003). These authors developed chloroplast DNA markers to characterize haplotype composition of degraded white oak DNA obtained from wood samples. This technique confirmed the correspondence between the genotype obtained



from living buds and that of timber collected from the same individual oak tree. A statistical procedure was used to test whether the haplotype of several wood samples was consistent with their presumed geographical origin. While the study showed that the technique could not unambiguously identify wood products of unknown origin, it could link the genetic composition of wood samples with their region of alleged origin.

Genetic pollution

Plant invasions from agriculture, forestry and horticulture have today captured the attention of most national conservation agencies as well as international forestry, agricultural and conservation NGOs (Higgins and Richardson 1998). The spread of 'wildlings' from exotic species' plantations is a well-known occurrence (Raybould and Gray 1994), and the literature on the concepts and concerns associated with 'invasive exotics' is now extensive (Rhymer and Simberloff 1996; Daniels and Sheil 1999). While it is known that the introduction of exotic populations can lead to competitive exclusion (displacement) of native populations, genetic pollution via hybridization is less well known (Huxel 1999).

Genetic pollution occurs when diversity in a native population or species' gene pool is reduced and/or 'damaged' as a result of hybridization in such a way as to cause concern about the distinctiveness or future fitness of the gene pool in question. While it is recognized that the mixing of population (and sometimes species') gene pools through hybridization and consequent introgression is a natural evolutionary process in forest trees and in plants and animals generally (Arnold 1992), the term 'genetic pollution' usually indicates a situation where hybridization was initiated or significantly influenced by human activities.

In the last decade, genetic pollution via pollen dispersal and/or hybridization has been recognized as a major concern in conservation biology. The risk of escape of transgenes from genetically modified organisms has helped focus public attention on the effects on native gene pools of hybridization and introgression from exotic species, nonlocal provenances and highly selected genotypes.

Genetic pollution in forest tree species is reported for the California black walnut (*Juglans hindsii*), which is known to have hybridized with many congeners imported for commercial purposes (Rhymer and Simberloff 1996). Another well-known example is that of the native European black poplar (*Populus nigra* L.), once widespread and now being extirpated over large parts of western Europe. Its habitat is being reduced and its gene pool threatened by the large-scale presence of cultivated hybrids. Another example is the California sycamore (*Platanus racemosa*), currently disappearing from its native range via introgression with the London plane tree (*Platanus x acerifolia*; Rhymer and Simberloff 1996). Box 7 describes research in Australia that was motivated by genetic contamination of native gene pools of *Eucalyptus* spp.

4. Concluding remarks

In most cases, conservation of FGRs is one aspect of a more complex management goal associated with sustainable forest management; a concept whose objectives alone can be difficult to determine, much less assess and measure. In this chapter, we presented the briefest of overviews of the practical usefulness and application of genetic research for FGR conservation and management.

The portfolio of examples presented shows how genetic research is attempting to address some very urgent forest conservation and management issues. As we saw, developing what we think might be solutions to a myriad of pressing problems, applying them, and then monitoring their effects are very significant challenges for the forest genetics research community. In part, this is because it is usually time-consuming and costly just to acquire good genetic data, interpret them and develop recommendations for the conservation and sustainable management of trees, their populations and their ecosystems; and that is assuming that we have asked the right questions in the first place and that there are decision-makers prepared to apply our recommendations in the second place. To a certain degree, forest geneticists presuppose that genetic data are all that are needed to conserve and manage FGRs effectively. But ecological, socioeconomic and even

Box 7. Genetic pollution of eucalyptus in Australia

Virtually all of the 800-or-so eucalyptus species (included in the genera *Angophora*, *Corymbia* and *Eucalyptus*; Hopper 1997) are endemic to Australia and the surrounding islands, and Australia has recognized that it has an international obligation to ensure the long-term management of this native genetic resource. Among Australia's indicators for sustainable forestry is the development of national strategies to manage gene pools of major plantation species and to minimize the impact of genetic pollution (Ministerial Council on Forestry 1997).

A project funded by the Australian Joint Venture Agroforestry Programme (JVAP), and supported by three research and development (R&D) corporations – Rural Industries, Land and Water Australia, and Forest and Wood Products – has issued a nationwide report on the risks and potential effects of genetic pollution from nonlocal *Eucalyptus* and hybrids on local eucalyptus species and populations. The report presents a framework for risk assessment and identifies key research topics that must be addressed for successful eucalyptus genetic management to occur. The report is part of an Agroforestry and Farm Forestry R&D programme, and aims to integrate sustainable and productive agroforestry into Australian farming systems. The principal research topics identified are:

- Distance and level of pollen dispersal
- Compatibility of plantation species with local native populations
- Potential of selecting for sustainability traits
- Spatial patterns of genetic variation within native gene pools of the most important plantation species
- Baseline molecular genetics for the long-term monitoring of introgression
- Impact of plantings on local diversity (Potts *et al.* 2001).

political (and other) factors play roles as important if not sometimes much more important than genetics in successful FGR conservation and management (e.g., understanding the causes of pollinator collapse, setting thresholds for sustainable resource extraction, controlling access, etc.).

For example, when trying to develop management guidelines for NWFPs, socioeconomic issues must be considered to be as significant as genetic and ecological ones in many instances. When human population numbers are low, when access is restricted, and when the resource being extracted is abundant and reproducing well, NWFPs exploited for subsistence purposes within a traditional use regime have the potential to be harvested indefinitely and sustainably. However, when collecting becomes more aggressive in order to supply market demand, the collector base expands, traditional harvesting regulations break down, and problems other than genetic ones can become much more urgent. In addition, this scenario says nothing about the many places in the world where entire forests are being removed or have already been removed for timber extraction and land conversion, and where forest geneticists have had little chance to intervene. It is hard to develop an FGR conservation strategy based on genetic considerations when there's nothing left to conserve. Indeed, there is empirical evidence that few wild resources can be commercially exploited in unregulated manners without ultimately facing extinction (Wilkie and Godoy 1996).

FGRs in the form of genetic diversity are also much less visible than the forests and the trees themselves, and it is therefore often much harder to capture the attention or the imagination of policy-makers when the adoption of FGR conservation strategies are recommended. Even though most natural resource planners would recognize genetic processes as essential components of ecosystem, species and population stability and adaptability, this is often an implicit assumption that is much harder to translate into targeted decisions, intervention planning and applied effort. Frequently, policies that are

inconsistent, too simple and/or just plain bad arise as a means of satisfying dominant or competing interest groups, and these can pose additional challenges to achieving FGR conservation objectives.

All the elements mentioned thus far concur to demonstrate that long-term sustainable management of FGRs will be a challenge for the forest genetic research community. But there appears to be agreement on the need to continue gathering accurate baseline genetic information on a broad range of forest tree species, to extrapolate findings from model species to others, to develop management recommendations based on genetic information, to integrate FGR conservation and sustainable use recommendations into acceptable socioeconomic frameworks, to raise public awareness, and to establish an effective dialogue with policy-makers.

References

- Aizen, M.A. and P. Feinsinger. 1994. Forest fragmentation, pollination and plant reproduction in a Chaco dry forest, Argentina. *Ecology* 75:330-351.
- Aldrich, P.R. and J.L. Hamrick. 1998. Reproductive dominance of pasture trees in a fragmented tropical forest mosaic. *Science* 281:103-105.
- Alia, R., J. Moro and J.B. Denis. 1997. Performance of *Pinus pinaster* provenances in Spain: interpretation of the genotype by environmental interaction. *Can. J. For. Res.* 27:1548-1559.
- Allard, R.W. 1970. Population structure and sampling methods. Pp. 97-107 *in* Genetic Resources in Plants – Their Exploration and Conservation (O.H. Frankle and E. Bennet, eds.). Blackwell Scientific Publications, Oxford, UK.
- Arnold, M.L. 1992. Natural hybridization as an evolutionary process. *Annu. Rev. Ecol. Syst.* 23:237-261.
- Baradat, P., M. Maillart, A. Marpeau, M.F. Slak, A. Yani and P. Pastiszka. 1995. Utility of terpenes to assess population structure and mating patterns in conifers. Pp. 5-27 *in* Population Genetics and Genetic Conservation of Forest Trees (P. Baradat, W.T. Adams and G. Müller-Starck, eds.). SPB Academic Publishing, Amsterdam, The Netherlands.
- Barret, S.C.H. and J. Kohn. 1991. Genetic and evolutionary consequences of small population size in plants: implications for conservation. Pp. 3-30 *in* Genetics and Conservation of Rare Plants (D.A. Falk and K.E. Holsinger, eds). Oxford University Press, New York, USA.
- Boshier, D.H. 2000. Mating Systems. Pp. 63-79 *in* Forest Conservation Genetics: Principles and Practice (A. Young, D.H. Boshier and T.J. Boyle, eds.). CSIRO publishing, Collingwood, Australia and CAB International, Wallingford, UK.
- Boyle, T.J. 2000. Criteria and indicators for the conservation of genetic diversity. Pp. 239-252 *in* Forest Conservation Genetics: Principles and Practice (A. Young, D.H. Boshier and T.J. Boyle, eds.). CSIRO publishing, Collingwood, Australia and CAB International, Wallingford, UK.
- Brown, A.H.D. and C.M. Hardner. 2000. Sampling the gene pools of forest tree species for *ex situ* conservation. Pp. 185-196 *in* Forest Conservation Genetics: Principles and Practice (A. Young, D.H. Boshier and T.J. Boyle, eds.). CSIRO publishing, Collingwood, Australia and CAB International, Wallingford, UK.
- Brown, A.H.D., A.G. Young, J.J. Burdon, L. Christidis, G. Clarke, D. Coates and W. Sherwin. 2000. Genetic indicators for state of the environment reporting. Department of Environment, Sports and Territories Technical Report, Canberra, Australia.
- Brown, S. and A.E. Lugo. 1994. Rehabilitation of tropical lands: a key to sustaining development. *Restor. Ecol.* 2:97-111.
- Castañeda, F. 2000. Criteria and indicators for sustainable forest management: international processes, current status and the way ahead. *Unasylva* 51(203):24-40.
- Castañeda, F., C. Palmberg-Lerche and P. Vuorinen. 2001. Criteria and indicators for sustainable forest management: a compendium. Forest Management Working Paper No 5, Forest Resources Development Service, Forest Resources Division, Food and Agriculture Organization of the United Nations (FAO), Rome, Italy.

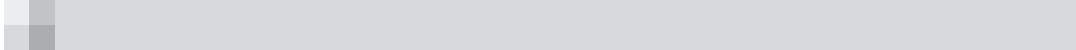
- Chase, M.R., R. Kesseli and K.S. Bawa. 1996. Microsatellite markers for population and conservation genetics of tropical trees. *Am. J. Bot.* 83:51-57.
- Conkle, M.T. 1997. Zonificación de semillas en México. Pp. 67-88 *in* Manejo de Recursos Genéticos Forestales (H. Vargas, J. Jesus, B. Bermejo-Velasquez and F.T. Ledig, eds.). Colegio de Postgraduados, Montecillo, México y División de Ciencias Forestales, Universidad Autónoma Chapingo, Chapingo, México.
- Crossa, J. 1989. Methodologies for estimating the sample size required for genetic conservation of outbreeding crops. *Theor. Appl. Genet.* 77:153-161.
- Crozier, R.H. 1997. Preserving the information content of species: genetic diversity, phylogeny and conservation worth. *Ann. Rev. Ecol. Syst.* 28:243-268.
- Dale, V.H., L.A. Joyce, S. McNulty, R.P. Neilson, M.P. Ayres, M.D. Flannigan, P.J. Hanson, L.C. Irland, A.E. Lugo, C.J. Peterson, D. Simberloff, F.J. Swanson, B.J. Stocks and B.M. Wotton. 2001. Climate change and forest disturbances. *Bioscience* 51(9):723-734.
- Daniels, R.E. and J. Sheil. 1999. Genetic pollution, concepts, concerns and transgenic crops. Pp. 65-72 *in* Gene Flow and Agriculture: Relevance to Transgenic Crops. BCPC symposium proceedings No. 72 (P.J.V. Lutwan, ed.). British Crop Protection Council (BCPC), Farnham, UK.
- Darwin, C. 1985. The effects of cross- and self-fertilization in the vegetable kingdom. John Murray, London, UK.
- Dawson, I. and J. Were. 1997. Collecting germplasm from trees – some guidelines. *Agrofor. Today* 9:6-9.
- Dawson, I.K., R. Waugh, A.J. Simons and W. Powell. 1997. Simple sequence repeats provide a direct estimate of pollen-mediated gene dispersal in the tropical tree *Gliricidia sepium*. *Mol. Ecol.* 6:179-183.
- Dayanandan, S., J. Dole, K.S. Bawa and R. Kesseli. 1999. Population structure delineated with microsatellite markers in fragmented populations of a tropical tree, *Carapa guianensis* (Meliaceae). *Mol. Ecol.* 8:1585-1593.
- Deguilloux, M.F., M.H. Pemonge, L. Bertel, A. Kremer and R.J. Petit. 2003. Checking the geographical origin of oak wood: molecular and statistical tools. *Mol. Ecol.* 12(6):1629-1636.
- Diamond, J.M. 1984. 'Normal' extinctions of isolated populations. Pp. 191-246 *in* Extinctions (M.H. Nitecki, ed.). University of Chicago Press, Chicago, USA.
- Dick, C.W., K. Abdul-Salim and E. Bermingham. 2003. Molecular systematics reveals cryptic Tertiary diversification of a widespread tropical rainforest tree. *Am. Nat.* 160(12):691-703.
- Ennos, R., R. Worrell and D.C. Malcolm. 1998. The genetic management of native species in Scotland. *Forestry* 71(1):1-23.
- Enquist, C.A.F. 2002. Predicted regional impacts of climate changes on the geographical distribution and diversity of tropical forests in Costa Rica. *J. Biogeogr.* 29:519-534.
- Eriksson, G., G. Namkoong and J.H. Roberds. 1993. Dynamic gene conservation for uncertain futures. *For. Ecol. Manage.* 62:15-37.
- Falconer, D.S. and T.F.C. Mackay (eds.). 1996. Introduction to quantitative genetics (4th edn.). John Wiley, New York, USA.
- Finkeldey, R. and H.H. Hatterer. 1993. Gene resources and gene conservation with emphasis on tropical forests. FAO/IBPGR (Food and Agriculture Organization/ International Board for Plant Genetic Resources) Plant Genetic Resources Newsletter 94/95:5-10.
- Foufopoulos, J. and A.R. Ives. 1999. Reptile extinctions on land-bridge islands: life-history attributes and vulnerability to extinction. *Am. Nat.* 153:1-25.
- Frankham, R. 1995. Inbreeding and extinction: A threshold effect. *Conserv. Bio.* 9:792-799.
- Frankham, R. 1996. Relationship of genetic variation to populations size in wildlife. *Conserv. Biol.* 10:1500-1508.
- Franklin, I.R. 1980. Evolutionary change in small populations. Pp. 135-149 *in* Conservation Biology: An Evolutionary–Ecological Perspective (M.E. Soulé and B.A. Wilcox, eds.). Sinauer Associates, Sunderland, USA.

- Funston, M. 1995. An approach to working with sustainability. USDA Forest Service, Sustainable Development Interdeputy Area Team, Discussion Paper 4. Downloaded from http://www.fs.fed.us/land/sustain_dev/susdev4.html.
- Gaston, K. J. 1994. Rarity. Chapman and Hall, London, UK.
- Gilpin, M.E. and M.E. Soulé. 1986. Minimum viable populations: approaches to modelling spatial structure. Pp. 11-27 *in* Metapopulations and Wildlife Conservation (D.R McCullough, ed.). Island Press, Washington, DC, USA.
- Gregorius, H.-R. 1991. Gene conservation and the preservation of adaptability. Pp. 31-47 *in* Species Conservation: A Population-Biological Approach (A. Seitz and V. Loeschke, eds.). Birkhäuser Verlag, Basel, Switzerland.
- Guariguata, M.R. and M.A. Pinard. 1998. Ecological knowledge of regeneration from seed in neotropical forest trees: Implications for natural forest management. *For. Ecol. Manage.* 112:87-99.
- Hammond, D.S., S. Gourlet-Fleury, P. van der Hout, H. ter Steege and V.K. Brown. 1996. A compilation of known Guianan timber trees and the significance of their dispersal mode, seed size and taxonomic affinity to tropical rain forest management. *For. Ecol. Manage.* 83:99-116.
- Hamrick J.L. 1992. Distribution of genetic diversity in tropical tree populations: implication for the conservation of genetic resources. Pp. 74-82 *in* Resolving Tropical Forest Resource Concerns Through Tree Improvement, Gene Conservation and Domestication of New Species (C.C. Lambeth and W. Dvorak, eds.). North Carolina State University, Raleigh, USA.
- Hamrick, J.L. and M.D. Loveless. 1989. The genetic structure of tropical tree populations: associations with reproductive biology. Pp. 19-146 *in* The Evolutionary Ecology of Plants (J.H. Bock and Y.B. Linhart, eds.). Westview Press, Boulder, USA.
- Hattemer, H.H. 1995. Concepts and requirements in the conservation of forest genetic resources. *For. Genet.* 2:125-134.
- Henle, K., K.F. Davies, M. Kleyer, C. Margules and J. Settele. 2004. Predictors of species sensitivity to fragmentation. *Biodivers. Conserv.* 13:207-251.
- Higgins, S.I. and D.M. Richardson. 1998. Pine invasions in the southern hemisphere: modelling interactions between organism, environment and disturbance. *Plant Ecol.* 135:79-93.
- Hijmans, R.J., L. Guarino, M. Cruz and E. Rojas. 2001. Computer tools for spatial analysis of plant genetic resources data: 1. DIVA_GIS. *Plant Genetic Resources Newsletter*, 127:15-19.
- Holsinger, K.E. and P. Vitt. 1997. The future of conservation biology: what is a geneticist to do? Pp. 202-216 *in* The Ecological Basis for Conservation: Heterogeneity, Ecosystems and Biodiversity (S.T.A. Pickett, R.S. Ostfeld, M. Shachack and G.E. Likens, eds.). Chapman and Hall, New York, USA.
- Hopper, S.D. 1997. An Australian perspective on plant conservation biology in practice. Pp. 255-278 *in* Conservation Biology for the Coming Decade (P.L. Fielder and P.M. Kareiva, eds.). Chapman and Hall, New York, USA.
- Hufford, K.M. and S.J. Mazer. 2003. Plant ecotypes: genetic differentiation in the age of ecological restoration. *Trends Ecol. Evol.* 18(3):147-155.
- Huxel, G.R. 1999. Rapid displacement of native species by invasive species: effects of hybridization. *Biol. Conserv.* 89:143-152.
- ITTO (International Tropical Timber Organization). 1998. Criteria and indicators for sustainable management of natural tropical forests. Policy Development Series No. 7. ITTO.
- IUCN. 2000. The 2000 IUCN red list of threatened species. <http://www.redlist.org>.
- Jennings, S.B., N.D. Brown, D.H. Boshier, T.C. Whitmore and J. do C.A. Lopes. 2001. Ecology provides a pragmatic solution to maintenance of genetic diversity in sustainably managed tropical rain forests. *For. Ecol. Manage.* 154:1-10.
- Johnson, N. and B. Cabarle. 1993. Surviving the cut: natural forest management in the Humid Tropics. World Resources Institute.
- Kim, Z-S. and H.H. Hattemer. 1994. Conservation and manipulation of genetic resources in forestry. Kwang Moon Kag Publishing Company, Seoul, Republic of Korea.

- Koski, V., T. Skrøppa, L. Paule, H. Wolf and J. Turok. 1997. Technical guidelines for genetic conservation of Norway spruce (*Picea abies* (L.) Karts.). International Plant Genetic Resources Institute (IPGRI), Rome, Italy.
- Kremer, A. 1994. Diversité génétique et variabilité des caractères phénotypiques chez les arbres forestiers. *Genetics, Selection, Evolution* 26(1):105s-123s.
- Krusche, D. and T. Geburek. 1991. Conservation of forest gene resources as related to sample size. *For. Ecol. Manage.* 40:145-150.
- Lamb, D. 1994. Reforestation of degraded tropical forest lands in the Asia-Pacific region. *J. Trop. For. Sci.* 7:1-7.
- Lamb, D., J. Parrotta, R. Keenan and N. Tucker. 1997. Rejoining habitat fragments: restoring degraded forest lands. Pp. 366-385 in *Tropical Forest Remnants: Ecology, Management and Conservation of 7 Forest Restoration Research Units* (W.F. Laurance and R.O. Bierregaard, eds.). University of Chicago Press, Chicago, USA.
- Lande, R. 1988. Genetics and demography in biological conservation. *Science* 241:1455-1460.
- Lande, R. 1995. Mutation and conservation. *Conserv. Biol.* 9:782-791.
- Laurance, W.F. 1991. Ecological correlates of extinction proneness in Australian tropical rainforest mammals. *Conserv. Biol.* 5:80-89.
- Lawrence, M.J., D.F. Marshall and P. Davies. 1995. Genetics of genetic conservation. II. Sample size when collecting seed of cross-pollinating species and the information that can be obtained from the evaluation of material held in genebanks. *Euphytica* 84:101-107.
- Leberg, P.L. 1993. Strategies for population reintroduction: effects of genetic variability on population growth and size. *Conserv. Biol.* 7:194-199.
- Ledig, F.T. 1986. Conservation strategies for forest gene resources. *For. Ecol. Manage.* 14:77-90.
- Lefèvre, F., N. Barsoum, B. Heinze, D. Kajba, P. Rotach, S.M.G. de Vries and J. Turok. 2001. EUFORGEN Technical Bulletin: *In situ* conservation of *Populus nigra*. International Plant Genetic Resources Institute (IPGRI), Rome, Italy.
- Lesica, P. and F.W. Allendorf. 1995. When are peripheral populations valuable for conservation? *Conserv. Biol.* 9:753-760.
- Levin, D.A. and H.W. Kerster. 1974. Gene flow in seed plants. *Evol. Biol.* 7:139-220.
- Luikart, G.L. and P.R. England. 1999. Statistical analysis of microsatellite DNA data. *Trends Ecol. Evol.* 14:253-255.
- Lynch, M. 1995. A quantitative genetic perspective on conservation issues. Pp. 471-501 in *Conservation Genetics: A Case History from Nature* (J.C. Avise and J.L. Hamrick, eds.). Springer-Verlag, New York, USA.
- Lynch, M., M. Pfreder, K. Spitze, N. Lehman, J. Hicks, D. Allen, L. Latta, M. Ottene, F. Bogue and J. Colbourne. 1999. The quantitative and molecular genetic architecture of a subdivided species. *Evolution*, 53:100-110.
- Marshall, D.R. and A.H.D. Brown. 1975. Optimum sampling strategies in genetic conservation. In *Crop Genetic Resources for Today and Tomorrow*. (O.H. Frankel and J.G. Hawkes, eds.). Cambridge University Press, Cambridge, UK.
- Martini, A., N.D. Rosa and C. Uhl. 1994. An attempt to predict which Amazonian tree species may be threatened by logging activities. *Environ. Conserv.* 21:152-162.
- Matyas, C. 1996. Climatic adaptation of trees: rediscovering provenance tests. *Euphytica* 92:45-54.
- McKinnell, F.H. 2002. Status and trends in indicators of forest genetic diversity. Forest Genetic Resources Working Paper no 38, Food and Agriculture Organization of the United Nations (FAO), Rome, Italy.
- McKinney, M.L. 1997. Extinction vulnerability and selectivity: combining ecological and paleontological views. *Annu. Rev. Ecol. Sys.* 28:495-516.
- McLachlan, J.S. and J.S. Clark. 2004. Reconstructing historical ranges with fossil data at continental scales. *For. Ecol. Manage.* 197:139-147.
- Mikola, J. 1982. Bud-set phenology as an indicator of climatic adaptation of Scots pine in Finland. *Silva Fennica* 16:178-184.

- Ministerial Council of Forestry, Fisheries and Aquaculture. 1997. Plantations for Australia: the 2002 Vision. No. 24, p. 88, Ministerial Council on Forestry, Fisheries and Aquaculture. Plantation 2020 Vision Implementation Committee, Canberra, Australia.
- Namkoong, G. 1988. Sampling for germplasm collections. *Hortscience* 23:79-81.
- Namkoong, G. 2001. Forest genetics: pattern and complexity. *Can. J. For. Res.* 31(4):623-632.
- Namkoong, G., T. Boyle, H-R. Gregorius, H. Joly, O. Savolainen, W. Ratnam and A. Young. 1997. Testing criteria and indicators for assessing the sustainability of forest management: genetic criteria and indicators. Working Paper No. 10, CIFOR (Centre for International Forest Research), Bogor, Indonesia.
- Namkoong, G., T. Boyle, Y.A. El-Kassaby, C. Palmberg-Lerche, G. Eriksson, H.-R. Gregorius, H. Joly, A. Kremer, O. Savolainen, R. Wickneswari, A. Young, M. Zeh-Nlo and R. Prabhu. 2002. Criteria and indicators for sustainable forest management: assessment and monitoring of genetic variation. Forest Genetic Resources Working Papers FGR/37E. Forest Resources Division, Food and Agriculture Organization of the United Nations (FAO), Rome, Italy.
- Nason, J.D., E.A. Herre and J.L. Hamrick. 1998. The breeding structure of a tropical keystone plant resource. *Nature* 391:685-687.
- Noss, R.F. 1983. A regional landscape approach to maintain diversity. *BioScience* 33:700-706.
- O'Neill G.A., I. Dawson, C. Sotelo-Montes, L. Guarino, M. Guariguata, D. Current and J.C. Weber. 2001. Strategies for genetic conservation of trees in the Peruvian Amazon. *Biodivers. Conserv.* 10: 837-850.
- Ouédraogo, A.S. and J.M. Boffa (eds.). 1999. Vers une Approche Régionale des Ressources Génétiques Forestières en Afrique sub-Saharienne. International Plant Genetic Resources Institute (IPGRI), Rome, Italy.
- Palmberg-Lerche, C. and S. Hald. 1999. Management of forest genetic resources: status and challenges. *Unasylva* 203(51):27-33.
- Parrotta, J. 1995. Influence of overstorey composition on understorey colonization by native species in plantations on a degraded tropical site. *J. Veget. Sci.* 6:627-636.
- Pederick, L.A. 1979. Natural variation in shining gum (*Eucalyptus nitens*). *Aust. For. Res.* 9:41-63.
- Petit, R.J., A. El-Mousadik and O. Pons. 1997. Identifying populations for conservation on the basis of genetic markers. *Conserv. Biol.* 12:844-855
- Pierce, A.R. and J.B. Ervin. 1999. Can independent forest management certification incorporate elements of landscape ecology? *Unasylva* 196(50):49-56
- Pinard, M.A., F.E. Putz, D. Rumiz, R. Guzman and A. Jardim. 1999. Ecological characterization of tree species for guiding forest management decisions in seasonally dry forests in Lomerio, Bolivia. *For. Ecol. Manage.* 113:201-213.
- Polansky, S., B. Csuti, C.A. Vossler and S.M. Meyers. 2001. A comparison of taxonomic distinctness versus richness as criteria for setting conservation priorities for North American birds. *Biol. Conserv.* 97:99-105.
- Potts, B.M., R.C. Barbour and A.B. Hingston. 2001. Genetic pollution from farm forestry using eucalypt species and hybrids. Report to RIRDC/LWA/FWPRDC (Rural Industries Research and Development Corporation/Land and Water Australia/Forest and Wood Products Research and Development Corporation) Joint Venture Agroforestry Programme. RIRDC publication no 01/114, RIRDC project no CPF-3A.
- Purvis, A., P.-M. Agapow, J.L. Gittleman and G.M. Mace. 2000. Nonrandom extinction and the loss of evolutionary history. *Science* 288:328-330.
- Rao, A.N. and V. Ramanatha Rao (eds.). 1999a. Bamboo and rattan genetic resources and use and research reports. IPGRI-APO (International Plant Genetic Resources Institute-Asia, the Pacific and Oceania), Serdang, Malaysia.
- Rao, A.N. and V. Ramanatha Rao (eds.). 1999b. Bamboo – Conservation, Diversity, Ecogeography, Germplasm, Resource Utilization and Taxonomy. IPGRI-APO (International Plant Genetic Resources Institute-Asia, the Pacific and Oceania), Serdang, Malaysia.

- Raup, D.M. 1994. The role of extinction in evolution. *Proc. Nat. Acad. Sci. USA* 91:6758-6763.
- Raybould, A.F. and A.J. Gray. 1994. Will hybrids of genetically modified crops invade natural communities? *Trends Ecol. Evol.* 9:85-89.
- Redford, K.H., P. Coppolillo, E.W. Sanderson, G.A.B da Fonseca, E. Dinerstein, C. Groves, G. Mace, S. Maginnis, R.A. Mittermeier, R. Noss, D. Olson, J.G. Robinson, A. Vedder and M. Wright. 2003. Mapping the conservation landscape. *Conserv. Biol.* 17:116-131.
- Rhymer, J.M. and D. Simberloff. 1996. Extinction by hybridization and introgression. *Annu. Rev. Ecol. Sys.* 27:83-109.
- Ricklefs, R.E. and D. Schluter. 1993. Species diversity: regional and historical influences. Pp 350-363 *in* Species Diversity in Ecological Communities. Historical and Geographical Perspectives (R.E. Ricklefs and D. Schluter, eds.). University of Chicago Press, Chicago, USA.
- Rocha, O.J.R. and G. Aguilar. 2001. Variation in the breeding behaviour of the dry forest tree *Enterolobium cyclocarpum* (Guanacaste) in Costa Rica. *Am. J. Bot.* 89:1600-1606.
- Rodrigues, A.S.L. and K.J. Gaston. 2002. Maximizing phylogenetic diversity in the selection of networks of conservation areas. *Biol. Conserv.* 105:103-111.
- Russell, G.J., T.M. Brooks, M.M. McKinney and C.G. Anderson. 1998. Present and future taxonomic selectivity in bird and mammal extinctions. *Conserv. Biol.* 12:1365-1376.
- Scherm, H. 2004. Climate change: can we predict the impacts on plant pathology and pest management? *Can. J. Plant Pathol.* 26(3):267-273.
- Sheil, D. 1999. Tropical forest diversity, environmental change and species augmentation: After the intermediate disturbance hypothesis. *J. Veget. Sci.* 10:851-860.
- Sherwin, W.B. and C. Moritz. 2000. Managing and monitoring genetic erosion. Pp. 9-34 *in* Genetics, Demography and viability of Fragmented Populations (A.G. Young and G.M. Clarke, eds.). Cambridge University Press, Cambridge, UK.
- Smith, T.B. and R.K. Wayne. 1996. Molecular genetic approaches in conservation. Oxford University Press, New York, USA.
- Stalker, H.T. and C.E. Simpson. 1995. Germplasm resources in *Arachis*. Pp. 14-53 *in* Advances in Peanut Science (H.E. Pattee and H.T. Stalker, eds.). American Peanut Research and Education Society, Stillwater, USA.
- Stern, K. and L. Roche. 1974. Genetics of Forest Ecosystems. Chapman and Hall, London, UK.
- Stork, N.E., T.J.B. Boyle, V. Dale, H. Eeley, B. Finegan, M. Lawes, N. Manokaran, R. Prabhu and J. Soberon. 1997. Criteria and Indicators for Assessing the Sustainability of Forest Management: Conservation of Biodiversity. Centre for International Forest Research (CIFOR) working paper No. 17, Bogor, Indonesia.
- Wilkie, D.S. and R.A. Godoy. 1996. Trade, indigenous rainforest economies and biological diversity. Pp. 83-102 *in* Current Issues in Non-Timber Forest Products Research. (M. Ruiz Perez and J.E.M. Arnold, eds.). Centre for International Forestry Research (CIFOR), Bogor, Indonesia.
- Woodroffe, R. and J.R. Ginsberg. 1998. Edge effects and the extinction of populations inside protected areas. *Science* 280:2126-2128.
- Young, A.G., T. Boyle and A.H.D. Brown. 1996. The population genetic consequences of habitat fragmentation for plants. *Trends Ecol. Evol.* 2:413-418.



Final remarks

Conservation of biodiversity and strategies for reducing poverty

The question of whether poverty reduction can be combined with biodiversity conservation is part of a debate familiar to conservationists (Adams *et al.* 2004): environmental conservation versus economic development. During the 20th century, the dominant approach to development was to push for economic growth and assume that environmental problems and social justice could be sorted out later. However, using Ghana as the example, Brashares *et al.* (2004) have recently shown that programmes aimed at promoting economic development, food security and biodiversity conservation can have positive interrelated outcomes. As demand for forest goods and services continues to grow and forest managers, policy-makers and land use planners are asked to make more and more complex decisions about forest use and access issues, equity will increasingly need to be addressed. In most cases, trade-offs will be inevitable (Rose and Chapman 2002).

While the value of forests to world food security has been widely documented (Pimentel *et al.* 1997), the relationship between forest genetic diversity and livelihoods is more difficult to study and measure. Indeed, with many national and international organizations including conservation and sustainable use of tropical forests among their top environmental and developmental priorities, we possess in fact only very limited knowledge of the effects of deforestation, fragmentation and uncontrolled exploitation on forest genetic resources (FGRs). In addition, our understanding of most appropriate practices for *in situ* and *ex situ* conservation of tropical forest tree species is only partial at best. Much more research is needed to document and to understand the levels of diversity that are needed to ensure sustainable economic development in forest-dwelling communities that depend on forest resources.

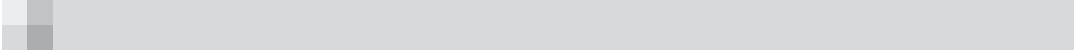
The complexity of biodiversity and of its sustainable use

Preserving, studying and using tropical forest diversity are challenges from conservation, research and utilitarian perspectives because intact natural forest ecosystems are rapidly being lost. This loss is not restricted just to tropical forests, for temperate and boreal forests are also affected, although deforestation and forest fragmentation are reducing the biodiversity of tropical forests at unprecedented rates.

Some authors argue that humans cannot make use of biodiversity without causing harm (Redford and Richter 1999) and the debate about the role of humans in nature, whether integral or separate, is ongoing (McNeely and Watchel 1988; Visser 1992). Nevertheless, there is now general acceptance that the conservation of forest biodiversity cannot be achieved solely within protected areas (Kemp 1992). Increased attention is being given to conservation values in the emerging definitions of sustainable forest management, to criteria and indicators for monitoring its achievement, and to participatory approaches, all of which point to increasing opportunities for strengthening biodiversity conservation in tropical forests under productive use.

However, biodiversity is complex and occurs at levels ranging from genes to species to populations to ecosystems and landscapes, each with its associated structure, composition and functional role in sustaining and reproducing itself. This implies that any biodiversity use pattern we choose to explore will have a cascading set of relationships and effects at the other levels. These relationships will not be easy to quantify or assess through single disciplines, nor will lone disciplines be able to master the spatial and temporal dimensions within which biodiversity operates and constantly changes.

Within the project presented in this volume, funded by the German Federal Ministry for Economic Cooperation and Development (BMZ), we tried to establish an interdisciplinary



and participatory framework that could simultaneously address genetic, ecological and socioeconomic issues of forest use in Argentina and Brazil. The objective was ultimately to devise methods and strategies for the conservation and sustainable use of FGRs that would guarantee the ongoing contribution of the forestry sector to the livelihoods of local communities and national economies.

Thus, we analyzed the effects of human activities, such as extraction of nonwood forest products (NWFPs), on FGRs with the goal of identifying management guidelines that could be applied locally and, hopefully at some point, more broadly. In fact, data from our research sites indicated that the sustainable use of NWFPs had the potential for playing a role in reducing deforestation rates and thus biodiversity losses. We also found that resource-use tenure, empowerment of local communities and good governance were key issues that underpinned the maintenance of forest cover in both countries (Chapters 6 and 8).

Indigenous management strategies tend to result in sustainable use of NWFPs only when a single cohesive community has sole access to the resource (Silva Matos and Bovi 2001). Nevertheless, we were encouraged by the fact that in more complex socioeconomic situations, collective action, strong social organization, good governance and secure tenure could contribute to achieving this goal, as the rubber tappers and the Mapuche communities have shown in Brazil (Chapter 8) and Argentina (Chapter 6), respectively. However, in such instances stricter penalties and monitoring may be required to control levels of exploitation and trade (Silva Matos and Bovi 2001).

However, the widely held argument that 'sustainable forestry' is the preferred option for tropical forest management has recently been criticized by people who argue that it is neither more profitable nor necessarily environmentally preferable to conventional logging (Pearce *et al.* 2002). They argue that the focus of sustainable forestry should shift from wood to nonmarket values such as environmental services, but that the return from these must exceed the returns of alternative land uses and the cost of management, including the cost of preventing entry by colonists. This argument demonstrates that until proper value is attributed to forest biodiversity and to its underlying genetic resources, and the importance of FGRs for forest-dwelling communities is recognized, both the livelihoods of these communities and their FGRs will continue to be threatened.

Thus, we support the ideas that forest management and sustainable use policies must include proper monetary and non-monetary returns to forest communities, that NWFPs should continue to be identified and their economic potential studied, and that stakeholders involved in forest management and use must be fully engaged in relevant decision-making processes. We also think that for it to be effective, conservation of FGRs must be integrated into the overall framework of sustainable forest management.

However, in many developing nations, the minimal conditions needed for this to occur have been hard to assemble, meaning that the sustainable management of forests seems to be less likely to emerge as a viable land-use option in the near term (Hyde 1999). Indeed, in some instances, the design and implementation of workable regulatory systems seem to be almost utopian, especially in countries that lack the political, administrative and enforcement frameworks within which such policies can be developed and implemented.

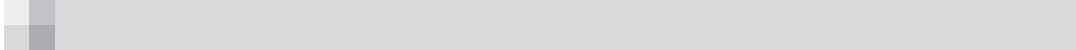
We therefore encourage the donor community and the relevant research institutions to continue to come together and to think strategically about how we can collectively engage and give a voice to forest-dwelling communities based on their very real needs. Equally importantly, we need to find ways to increase the likelihood that policy-makers in the lesser-resourced countries will support the conservation and sustainable use of FGRs in their countries.

Before regional conservation and sustainable use efforts can be truly meaningful, national FGR programmes must be put into operation and supported by policy-makers. The commitment of policy-makers to the conservation of forest genetic diversity can be increased only if they are aware of the potential of FGRs for development. Increasing the use of FGRs within sustainable use frameworks will lead to FGR conservation, improved stability and predictability within local resource use, better regional collaboration and common conservation actions for priority species.

The importance of these issues needs to be kept in the public eye, hence the decision to produce this book.

References

- Adams, W., R. Aveling, D. Brockington, B. Dickson, J. Elliott, J. Hutton, D. Roe, B. Vira and W. Wolmer. 2004. Biodiversity conservation and the eradication of poverty. *Science* 306:1146-1148.
- Brashares, J.S., P. Arcese, M.K. Sam, P.B. Coppolillo, A.R.E. Sinclair and A. Balmford. 2004. Bushmeat hunting, wildlife declines and fish supply in West Africa. *Science* 306:1180-1183.
- Hyde, W. 1999. Patterns of forest development. Lecture given at International Institute for Environment and Development, London, April 1999.
- Kemp, R.H. 1992. ITTO and the conservation of biological diversity. *In* *Conserving Biological Diversity in Managed Tropical Forests* (J.M. Blockhus, M. Dillenbeck, J.A. Sayer and P. Wegge, eds.). IUCN (The World Conservation Union), Cambridge, UK.
- McNeely, J.A. and P.S. Watchel. 1988. *Soul of the Tiger*. Oxford University Press, Oxford, UK.
- Pearce, D., F.E. Putz and J.K. Vanclay. 2001. Sustainable forestry in the tropics: panacea or folly? *For. Ecol. Manage.* 172:229-247.
- Pimentel D., M. McNair, L. Buck, M. Pimentel and J. Kamil. 1997. The value of forests to world food security. *Hum. Ecol.* 25(1):91-120.
- Redford, K.H. and B.D. Richter. 1999. Conservation of biodiversity in a world of use. *Conserv. Biol.* 13(6):1246-1256.
- Rose, S.K. and D. Chapman. 2002. Timber harvest adjacency economies, hunting, species protection and old growth value: seeking the dynamic optimum. *Ecol. Econ.* 44:325-344.
- Silva Matos, D.M. and M.L.A. Bovi. 2001. Understanding the threats to biological diversity in southeastern Brazil. *Biodivers. Conserv.* 11(10):1747-1758.
- Visser, M.B.H. 1992. Environmental ethics – a case for survival. Pp. 534-544 *in* *In Harmony with Nature* (S.K. Yap and S.W. Lee, eds.). Malayan Nature Society, Kuala Lumpur, Malaysia.





In the second half of the 20th century, development strategies tended to be narrowly focused on economic growth. Biodiversity, natural resource planning and social justice took a back seat to dollars and balance sheets. Often they were simply ignored. In South America, this myopic view of progress resulted in the massive destruction of tropical forests.

Removing large tracts of forest radically alters the physical landscape and destroys natural habitats. It also erodes the genetic diversity of plants, animals and other organisms – diversity that is vital for species' ability to adapt to new conditions. When biological wealth is undermined, so too are human livelihoods. Yet nearly half a decade into the 21st century, we still do not fully understand the impacts of pervasive deforestation.

This book, *Challenges in Managing Forest Genetic Resources for Livelihoods*, is a must-read for all who work directly or indirectly on forest conservation or rural development in forested areas. Based on a four-year project across four different forest ecosystems in South America, it examines how current patterns of use affect forest genetic resources, and considers the fate of the forests and those who depend on them. Above all, this book asks: can we successfully combine poverty reduction with biodiversity conservation? What emerges is that while advances can and have been made, it is crucial that policy-makers and governments fully support the process.