THE STATE
OF THE WORLD'S
FOREST GENETIC RESOURCES
THEMATIC STUDY

GENETIC CONSIDERATIONS IN ECOSYSTEM RESTORATION USING NATIVE TREE SPECIES

Food and Agriculture Organization of the United Nations

Bioversity International

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GENETIC CONSIDERATIONS IN ECOSYSTEM RESTORATION USING NATIVE TREE SPECIES

Editors

Michele Bozzano,[1] Riina Jalonen,[1] Evert Thomas,[1] David Boshier,[1,2] Leonardo Gallo,[1,3] Stephen Cavers,[4] Sándor Bordács,[5] Paul Smith[6] and Judy Loo[1]

[1] Bioversity International, Italy
[2] Department of Plant Sciences, University of Oxford, United Kingdom
[3] Unidad de Genética Ecológica y Mejoramiento Forestal, INTA Bariloche, Argentina
[4] Centre for Ecology and Hydrology, Natural Environment Research Council, United Kingdom
[5] Central Agricultural Office, Department of Forest and Biomass Reproductive Material, Hungary
[6] Seed Conservation Department, Royal Botanic Gardens, Kew, United Kingdom

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Foreword

One of the major and growing environmental challenges of the 21st century will be the rehabilitation and restoration of forests and degraded lands. Notwithstanding the large-scale restoration projects initiated in Africa and Asia as of the 1970s, the current level of interest in forest and landscape restoration is more recent. With the adoption of the strategic plan of the United Nations Convention on Biological Diversity for 2011-2020, a strong new impetus has been given not only to halt degradation, but to reverse it. The plan states that, by 2020, 15 percent of all degraded lands should be restored. This target is consistent with the Bonn Challenge, which calls for restoring 150 million hectares of degraded land by 2020.

Forests play a crucial part in resilient landscapes at multiple scales. Restoring forest ecosystems is therefore a key strategy not only for tackling climate change, biodiversity loss and desertification, but can also yield products and services that support local people's livelihoods.

Restoration is not only about planting trees. Its success requires careful planning, as painfully demonstrated by numerous past restoration projects that have not attained expected goals. Restoration practices must be based on scientific knowledge, particularly so in these times of progressive climate change. The trees we plant today and other associated measures for restoration and rehabilitation of degraded ecosystems must be able to survive abiotic and biotic pressures, including social ones, in order to be self-sustaining and generate the products and services vital to supporting the world's population and environment for the years to come.

Biodiversity International coordinated this thematic study as an input to FAO's landmark report on *The State of the World's Forest Genetic Resources*. The report was requested by the Commission on Genetic Resources for Food and Agriculture, which guided its preparation, and agreed, in response to its findings, on strategic priorities which the FAO Conference adopted in June 2013 as the *Global Plan of Action for the Conservation, Sustainable Use and Development of Forest Genetic Resources*.

The publication of this study is an important step in the implementation of the *Global Plan of Action*. It provides fundamental information for the achievement of knowledge-based ecosystem restoration using native tree species. It draws attention to the importance of embedding genetic considerations in restoration activities, an aspect which is often overlooked both by restoration scientists and practitioners, but is nonetheless crucial to rebuilding resilient landscapes and ecosystems. We trust that it will contribute to informing future restoration efforts and help to ensure their success.

Eduardo Rojas-Briales
Assistant Director-General, Forestry Department
Food and Agriculture Organization of the United Nations

Stephan Weise
Deputy Director General – Research
Bioversity International

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Part 1

OVERVIEW

Chapter 1

Introduction

Evert Thomas,[1] Riina Jalonen,[1] Leonardo Gallo,[1,2] David Boshier[1,3] and Judy Loo[1]

[1] *Bioversity International, Italy*
[2] *Unidad de Genética Ecológica y Mejoramiento Forestal, INTA Bariloche, Argentina*
[3] *Department of Plant Sciences, University of Oxford, United Kingdom*

FAO (2010) estimates that 13 million hectares of natural forests are lost each year worldwide. This has been accompanied by an increase in the area reforested and of forested ecosystems restored. Between 2000 and 2010, almost 5 million hectares of trees were planted annually, an area equivalent to that of Costa Rica (FAO, 2010). It is estimated that 76 percent of this area was planted mainly for productive purposes and 24 percent for protective purposes, although planted forests in both categories may serve multiple purposes (FAO, 2006). Presumably, many trees were also planted in other types of landscape and production systems that were not included in these statistics, such as farmland, and for which little information is available on a global scale. The area of planted forests is expected to continue to increase, reaching 300 million hectares by 2020 (FAO, 2010). Examples of large-scale reforestation and forest restoration initiatives are listed in Table 1.1.

The global interest in planting trees holds significant promise for restoring degraded ecosystems, mitigating effects of environmental changes, conserving biodiversity, and yielding products and services that support local people's livelihoods. Globally, it is estimated that 2 billion hectares of land could benefit from restoration; this is an area larger than South America (WRI, 2011; Laestadius *et al.*, 2012). The ability of forest ecosystem restoration to mitigate the impacts of numerous environmental problems, and to slow and eventually reverse their negative effects, is widely recognized in international agreements, including the United Nations Framework Convention on Climate Change, the Convention on Biological Diversity, the United Nations Convention to Combat Desertification, the Aichi Biodiversity Targets[1] and the European Union Biodiversity Targets for 2020.[2] In particular, restoration and reforestation hold vast potential not only for mitigating the impacts of climate change, through sequestration of atmospheric carbon dioxide in plant biomass (Canadell and Rapauch, 2008; Alexander *et al.*, 2011a), but also for halting biodiversity loss and countering the encroachment of the arid frontier (see Insight 2).

In spite of serious concerns that restoration may become a new excuse for continued agribusiness exploitation and expanded industrial plantations of exotic tree species that are not likely to enhance biodiversity and ecosystem services or benefit local communities (Alexander *et al.*, 2011a), the growing global interest in reforestation and restoration is accompanied by an increasing interest in using native plant material (Rogers and Montalvo, 2004; Aronson *et al.*, 2011; Montagini and Finney, 2011; Newton and Tejedor, 2011; Lamb, 2012). However, an important concern in the shift to native species is the selection of appropriate genetic planting stocks for use in restoration activities (Rogers and Montalvo, 2004).

[1] http://www.cbd.int/sp/targets/

[2] http://www.cbd.int/nbsap/about/targets/eu

TABLE 1.1.
Examples of large-scale tree planting and forest landscape restoration initiatives (as of March 2012)

Initiative (year of initiation)	Scale	Country or region	Leading or coordinating institution
Green Belt Movement (1977)	45 million trees planted	Originating in Kenya, now a worldwide movement	Established by Professor Wangari Maathai
Green Wall of China (1978)	Planned to be 4500 km long and cover 35 million ha, of which it is estimated that two-thirds have been achieved so far	China, bordering the Gobi desert	Government of China
Great Green Wall (2005)	Planned to be a tree belt 15 km wide and 7775 km long, with an area of 11.7 million ha	Sahel across Africa, with 11 countries, from Senegal to Djibouti, participating	African Union
Billion Tree Campaign (2006)	12 billion trees planted	Global	United Nations Environment Programme, Plant for the Planet Foundation
The Atlantic Forest Restoration Pact (2009)	Aims to restore 15 million ha of degraded lands in the Brazilian Atlantic Forest biome by 2050, and to sustainably manage the remaining forest fragments	Brazilian Atlantic Forest biome	Joint effort of non-governmental organizations, the private sector, government and research institutions
The Green Mission (2010)	Plans to afforest or restore 5 million ha of degraded and cleared forests, and improve the quality of another 5 million ha over the next 10 years	India	Ministry of Environment and Forests
Aichi Nagoya Target 15 (2010)	Restoration of at least 15% of degraded ecosystems by 2020, as part of the target to enhance ecosystem resilience and the contribution of biodiversity to carbon stocks through conservation and restoration	Global	Parties to the Convention on Biological Diversity
Rwanda's Forest Landscape Restoration Initiative (2011)	Plans to restore forest nationwide "from border to border"	Rwanda	The Government of Rwanda in collaboration with the International Union for Conservation of Nature (IUCN), the Secretariat of the United Nations Forum on Forests and the private sector
The Bonn Challenge (2011)	Targets to restore 150 million ha of deforested and degraded lands	Global	Announced at the Bonn Challenge Ministerial Roundtable in September 2011; supported and promoted by IUCN, World Resources Institute and the Global Partnership on Forest Landscape Restoration, among others

In this thematic study we discuss the use of native species and genetic considerations in a selection of current approaches to ecosystem restoration, and identify the most important bottlenecks that currently restrict the generalized use of native species, and which may put at risk the long-term success of restoration efforts. Our main message is that increasing the use of native species in restoration activities provides real environmental and livelihood benefits, but also involves clear risks, mainly related to the selection of the appropriate genetic source for the target plant species.

First and foremost, increasing the use of native species in restoration activities contributes to conservation of the species themselves and their genetic diversity. Second, if planting material represents not only a native species but originates from seed sources local to the planting site, it will have evolved together with other native flora and fauna of the area. It should therefore be well adapted to cope with the local environment and should support native biodiversity and ecosystem resilience to a greater extent than would introduced (exotic) planting material (Tang *et al.*, 2007). Third, native species may be less likely either to become invasive or to succumb to introduced or native pests than exotic species (Ramanagouda *et al.*, 2010; Hulme, 2012). Finally, native species may correspond better to the preferences of local people, and chances are also higher that local people hold ethnobotanical and ethno-ecological knowledge of native species, which may facilitate their successful use in restoration projects (Shono, Cadaweng and Durst, 2007; Chazdon, 2008; Douterlungne *et al.*, 2010). In turn, promoting native species that produce non-timber forest products can contribute to the conservation of related traditional knowledge as well as the cultures that maintain it.

Use of exotic species in reforestation and forest restoration can result in negative impacts for conservation and the environment (Richardson, 1998; Pimentel, Zuniga and Morrison, 2005; Stinson *et al.*, 2006; Tang *et al.*, 2007; see also Insight 3: Invasive species and the inappropriate use of exotics). However, it must be recognized that the exotic versus native species debate is not free of controversy. There may be situations in which the benefits generated by exotics largely outweigh the disadvantages, not only in socioeconomic terms but also in ecological terms (D'Antonio and Meyerson, 2002; Alexander *et al.*, 2011a). In addition, it would be unrealistic to think that exotics can be completely eliminated from the environments in which they have been introduced and in some cases have become naturalized. Better understanding of local people's preferences can help promote the use of those exotics already introduced, with clear benefits for restoration projects. However, species with known invasive potential should be avoided.

It is not always easy to establish with certainty whether a species is native to a particular area or has been introduced by humans, possibly long ago (e.g. Vendramin *et al.*, 2008). Some exotic tree species – most notably *Eucalyptus* and *Pinus* spp. – have been deliberately introduced to various parts of the world for their perceived greater utility or production capacity, and because knowledge about their propagation is generally greater than that about native alternatives. The global spread of homogeneous planted forests, centred on eucalypts, pines and poplars, was largely driven by industry that had developed in areas where these species occurred naturally and had tailored its production lines to the wood properties of these species. In addition, the distribution of species (and provenances) by humans is often an outcome of unplanned events (Finkeldey, 2005).

It is clear that in the short term it will not be possible to replace the predominant use of exotics with use of native species for restoration and reforestation. Currently, most of the planted forests in the tropics still comprise exotic tree species selected mainly for their production functions. The proportion of exotic species in afforestation or reforestation initiatives between 2003 and 2007 was reported to be 82 percent in western and central Africa, 99 percent in eastern and southern Africa, 28 percent in East Asia, 94 percent in South and Southeast Asia and 98 percent in South America (calculated from FAO, 2010: 92). While there are probably hundreds of native species with growth performance and wood quality at least comparable to that of the commonly used plantation species, lack of knowledge about the biology, propagation and management of such native species is currently among the main constraints for their wider use (Newton, 2011; Lamb, 2012), along with the difficulties of trying to alter industrial systems tailored to particular production species. The time seems ripe now for large-scale investments to overcome these limitations.

Despite the expected benefits of using native species, increasing the scale of restoration activities will be associated with elevated risks of failure if some basic guidelines are not followed. For example, only two out of 98 publicly funded reforestation projects in Brazil were considered successful during an evaluation in 2000 (Wuethrich, 2007). Reforestation and restoration efforts may fail for a variety of reasons, from wrong species for wrong sites to inappropriate silvicultural approaches and techniques (Rogers and Montalvo, 2004; Le *et al.*, 2012). In general, little information is available about the global success of tree-planting efforts, especially in areas where ecosystems may be severely degraded or initial growing conditions are particularly harsh. People are often hesitant to share information on failures in spite of the help it could provide to improving current practices, and global efforts to record reforestation and forest restoration activities started only recently (FAO, 2010). However, the annual average area reported for afforestation and reforestation activities globally in 2003–07 was more than twice the annual average increase in the area of planted forests over the ten-year period 2000–2010 (FAO, 2010). Low success rates in establishment and survival of seedlings can be assumed to contribute to the difference.

Although the reasons for frequent failures in reforestation and restoration activities are not often known, it is probable that many failures are related to poor matching of planting material to the target site, or too narrow a genetic base for the planting stock (Rogers and Montalvo, 2004). Indeed, to attain a functional and resilient ecosystem, it is crucial that the genetically adapted planting material used for establishing a plant community represents a certain minimum level of intraspecific diversity to ensure that its progeny will in turn be viable and able to produce viable offspring. Aside from the initial quality and genetic diversity of germplasm, and its suitability for the planting site, the extent of gene flow across landscapes over subsequent generations is also of central importance for the successful long-term restoration of ecosystems and tree populations. This ensemble of genetic qualities is necessary not only to provide the desired forest functions, products and services, but also to enable restored populations to reproduce and survive on the site.

Genetic diversity has generally been found to be positively related not only with the fitness of individual plant populations (Reed and Frankham, 2003; Rogers and Montalvo, 2004), but also with the stability and resilience of ecosystems (Gregorius, 1996; Elmqvist *et al.*, 2003; Müller-Starck, Ziehe and Schubert, 2005; Thompson *et al.*, 2010; Sgro, Lowe and Hoffmann, 2011). Tree communities need particularly adaptive genetic variation to succeed over time on the restored site; such variation promotes survival and good growth while at the same time enhancing resilience and resistance to biotic and abiotic stresses such as environmental variations (Pautasso, 2009; Dawson *et al.*, 2011; Schueler *et al.*, 2012) or pests and pathogens (Schweitzer *et al.*, 2005; Cardinale *et al.*, 2012). In the long term, adaptive genetic diversity will promote successful reproduction, reduce the risk of inbreeding and genetic impoverishment that can result from genetic drift, and increase a population's ability to adapt to future site conditions.

Currently little is known about the genetic diversity of most native species, particularly the thousands of tropical tree species that could play an important role in restoring degraded tropical ecosystems and their functions. Where guidelines exist, for example on the collection of germplasm, they appear to be largely unknown or overlooked by restoration practitioners. Moreover, despite the high expectations for restored forests to mitigate climate change, ensuring the capability of tree populations to adapt to changing environment as a precondition for their mitigation function has received hardly any attention. The fact that the negative effects of genetic homogeneity are not necessarily immediately evident but accumulate over time means that resulting problems are difficult to perceive (Rogers and Montalvo, 2004) and address. Furthermore, by the time the effects are obvious they may already have affected large areas. For example, low genetic diversity in planting material, stemming from collecting seed from single isolated trees, can lead to increased homozygosity, particularly in the next generation, and may result in the expression of

Box 1.1.
Key concepts in ecosystem restoration

A **degraded ecosystem** "exhibits loss of biodiversity and a simplification or disruption in ecosystem structure, function and composition caused by activities or disturbances that are too frequent or severe to allow for natural regeneration or recovery" (Alexander *et al.*, 2011b).

Ecological restoration is "the process of assisting the recovery of an ecosystem that has been degraded, damaged, or destroyed" (SER, 2004). Alexander *et al.* (2011b) define ecological restoration as "an intentional activity that initiates or facilitates the recovery of ecosystems by re-establishing a beneficial trajectory of maturation that persists over time. The science and practice of ecological restoration is focused largely on reinstating autogenic ecological processes by which species populations can self-organize into functional and resilient communities that adapt to changing conditions while at the same time delivering vital ecosystem services. In addition to reinstating ecosystem function, ecological restoration also fosters the re-establishment of a healthy relationship between humans and their natural surroundings by reinforcing the inextricable link between nature and culture and emphasizing the important benefits that ecosystems provide to human communities."

Forest restoration aims to "restore the forest to its state before degradation (same function, structure and composition)" (ITTO, 2002).

Forest landscape restoration is "a planned process that aims to regain ecological integrity and enhance human wellbeing in deforested or degraded forest landscapes" (WWF and IUCN, 2001).

Rehabilitation is "a process to re-establish the productivity of some, but not necessarily all, of the plant and animal species thought to be originally present at a site. For ecological or economic reasons the new forest might also include species not originally present at the site. The protective function and many of the ecological services of the original forest may be re-established" (Gilmour, San and Xiong Tsechalicha, 2000).

Reforestation is "the re-establishment of forest through planting and/or deliberate seeding on land classified as forest, for instance after a fire, storm or following clearfelling" (FAO, 2010).

Afforestation is "the act of establishing forests through planting and/or deliberate seeding on land that is not classified as forest" (FAO, 2010).

Planted forests are forests "composed of trees established through planting and/or through deliberate seeding of native or introduced species" (FAO, 2010).

Resilience is "the ability of an ecosystem to recover from, or to resist stresses (e.g. drought, flood, fire or disease)" (Walker and Salt, 2006).

A **native species** (also indigenous species) is a species which is part of the original flora of an area (IBPGR, now Bioversity International).

An **exotic species** (also alien or introduced species) is "a species which is not native to the region in which it occurs" (FAO, 2002).

Naturalized species are "intentionally or unintentionally introduced species that have adapted to and reproduce successfully in their new environments" (FAO, 2002).

A **provenance** refers to "the original geographic source of seed, pollen or propagules" (FAO, 2002).

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Box 1.1. *(continued)*
Key concepts in ecosystem restoration *(continued)*

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deleterious recessive alleles, which in turn decreases individual fitness (i.e. inbreeding depression) (White, Adams and Neale, 2007). Inbreeding can have impacts at any stage of development, for example through reduced embryo viability, seedling survival, tree vigour or seed production (see Insight 1: Examples illustrating the importance of genetic considerations in ecosystem restoration).

Restoration, rehabilitation and reforestation are all terms commonly used to refer to re-establishing forest vegetation on deforested areas. In this study we use the term "ecosystem restoration." This largely coincides with "ecological restoration," defined as "the process of assisting the recovery of an ecosystem that has been degraded, damaged, or destroyed" (SER, 2004), but also aims to accommodate rehabilitation and reforestation activities that do not necessarily comply with some more conservative definitions of restoration (Lamb, 2012). These and other terms related to ecosystem restoration are defined in Box 1.1. We acknowledge that restoration is not the most appropriate term for characterizing some of the activities described in this and the following chapters because it suggests the aim of re-establishing a pre-existing ecosystem. In some cases it is almost impossible to define a previous state to which an ecosystem can be restored (Hilderbrand, Watts and Randle, 2005). It may also be impossible to return ecosystems to historical states because of radical changes that have already taken place (e.g. severe aridification, soil degradation or socioeconomic changes) (Buizer, Kurz and Ruthrof, 2012), or the objective of a restoration activity may simply be less ambitious with respect to the plant community it aims to establish (Lamb, 2012). In spite of these shortcomings, we have chosen to use "ecosystem restoration" throughout this study for the sake of uniformity.

While the systems and approaches discussed in this study cover a range of objectives and species assemblages, sometimes including exotic species, they all emphasize the use of indigenous tree species and diversity for their intrinsic relationships with indigenous flora and fauna and local knowledge and cultures.

1.1. Objectives and organization of the study

The objective of this thematic study is to review and analyse current practices in ecosystem restoration, with a particular focus on the use of native tree species and genetic considerations related to the selection of appropriate planting material. Based on this analysis we put forward a number of practical recommendations, including genetic considerations in ecosystem restoration, that are intended to help practitioners to avoid genetic problems and enhance both the short- and long-term success of future restoration activities. Our target audience includes researchers, restoration practitioners and policy-makers.

Box 1.2.
It's not just about restoring plants

While the emphasis here has been on sourcing seed for restoration, it is important to recognise that many species have intimate associations with a range of organisms and that these too may require restoration. The "If you build it, they will come" paradigm does not always apply and ill-considered placement of restoration projects can lead to poor utilization by the very organisms they are expected to attract to recreate interactions and processes at the population and community level. In addition, there can be considerable benefits for simultaneously restoring plants and associated organisms. For example, the survival and growth of acacias is significantly improved if seed is simultaneously planted with nitrogen-fixing bacterial symbionts, with excess nitrogen benefiting other co-planted species, resulting in a better and more rapid restoration outcome (Thrall *et al.,* 2005).

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This report is organized in five main parts, including this introduction. In the second part, experienced scientists briefly present theoretical and practical issues relevant to ecosystem restoration, with particular emphasis on genetic aspects. This more theoretical series of contributions serves as a basis for the analysis of the restoration methods and approaches and underpins the recommendations. The third part is an overview of various methods and approaches that are currently used in ecosystem restoration and are based – at least partially – on the use of native species. The authors contributing to the presentation of these methods and approaches were requested to reply to a set of questions aimed at facilitating an analysis of the methods they used and their genetic implications; the questionnaire is available on the Bioversity website.[3] The fourth part presents an analysis of the use of genetic considerations in current restoration methods, as well a number of action and research recommendations, building on the previous chapters of theoretical and general considerations, presentation of the methods and approaches, and the responses to the survey. The fifth and final part summarizes the main conclusions of this thematic study.

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Insight 1

Examples illustrating the importance of genetic considerations in ecosystem restoration

David Boshier,[1,3] Evert Thomas,[1] Riina Jalonen,[1] Leonardo Gallo[1,2] and Judy Loo[1]

[1] Bioversity International, Italy
[2] Unidad de Genética Ecológica y Mejoramiento Forestal, INTA Bariloche, Argentina
[3] Department of Plant Sciences, University of Oxford, United Kingdom

Poor genetic matching of planting material to the target site may result in reduced viability of restoration projects

The widespread and severe dieback in three ponderosa pine plantations planted south of Pagosa Springs, Colorado, United States, in the late 1960s to mid-1970s has been related to the use of inappropriate genetic seed source. A pathogen (*Cenangium ferruginosum*) has been identified in the plantations, but observations are consistent with this being a secondary impact and not the primary cause of failure (Worral, 2000; Rogers and Montalvo, 2004).

Use of provenance trials to guide genetic matching

The natural range of black walnut (*Juglans nigra* L.) extends from the eastern United States west to Kansas, South Dakota and eastern Texas. A subset of 15 to 25 sources from 66 sampled provenances was planted in each of seven geographically disparate common-garden field trials. After 22 years, survival was much higher for local trees (71 percent) than for the other provenances (zero survival at some sites) (Bresnan *et al.*, 1994; Rogers and Montalvo, 2004). This allowed the authors to make informed decisions about where best to use what germplasm.

Selfing (self-pollination) can considerably affect survival and size of offspring

In a study in which offspring of *Pseudotsuga menziesii* selfed and outcrossed crosses were compared 33 years after establishment of seedlings, the average survival of selfed offspring was only 39 percent that of the outcrossed individuals. Moreover, the average diameter at breast height (DBH) of the surviving selfed trees was 59 percent that of the surviving outcrossed siblings (White, Adams and Neale, 2007).

Low levels of genetic diversity can compromise successful mating between plant individuals

Attempts to restore the endangered daisy *Rutidosis leptorrhynchoides* were constrained by the limited reproductive potential of small populations (fewer than 200 plants) where the low number of self-incompatibility alleles prevented successful mating between many of the remnant plants (Young *et al.*, 2000). Among trees, several *Prunus* species are known to have self-incompatibility alleles, so the same considerations could apply.

Negative consequences of low genetic diversity of the source material usually accumulate in the subsequent generations

Acacia mangium was first introduced to Sabah (Malaysia) from Australia in 1967 in two small stands of 34 and approximately 300 trees of the "maternal half-sib family." This material formed the basis for more than 15 000 hectares of plantations. A simple nursery trial comparing seedlings from the first to third generation showed a reduced height growth in seedlings harvested from the second and third generation, as compared with the first generation (20.7 cm and 18.1 cm, compared with 32.5 cm) (Sim, 1984).

Selection for favourable characteristics can considerably improve the quality of individuals where specific objectives have been set for the planted forests

Tree improvement programmes have been successful in dramatically increasing growth and quality in commercially valuable and widely planted species. For example, a study compared the performance of *Acacia auriculiformis* trees grown from seedlots obtained from: (1) a seedling seed orchard (SSO), (2) a seed production area (SPA), (3) a natural-provenance site (NPS) and (4) a commercial seedlot from the same provenance (CS) from Viet Nam. Four-year old trees grown from the SSO and SPA seedlots scored significantly higher than trees from the NPS for a number of traits including height, DBH, conical stem volume, stem straightness and axis persistence. In contrast, trees grown from commercial seedlots scored consistently lower for these traits (Hai *et al.*, 2008). Inbreeding may have contributed to the poor growth and quality of trees originating from the commercial seedlots.

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Insight 2

The Great Green Wall for the Sahara and the Sahel Initiative: building resilient landscapes in African drylands

Nora Berrahmouni, François Tapsoba and Charles Jacques Berte

Forest Assessment, Management and Conservation Division, Forestry Department, Food and Agriculture Organization of the United Nations, Rome, Italy

Desertification,[4] land degradation and drought, combined with climate change, have a strong negative impact on the food security and livelihoods of local communities in Africa's drylands, home to some of the world's poorest populations.

The Great Green Wall for the Sahara and the Sahel Initiative (GGWSSI) was launched by African heads of state and government "to improve the resilience of human and natural systems in the Sahel–Saharan zone to Climate Change through a sound ecosystems' management, sustainable development of land resources, protection of rural heritage and improvement of the living conditions and livelihoods of populations living in these areas." This African Union initiative, based on a proposal of former President of Nigeria, H.E. Olusegun Obasanjo, involves over 20 countries bordering the Sahara.

The Food and Agriculture Organization of the United Nations (FAO), the European Union and the Global Mechanism of the UNCCD are supporting the African Union Commission and 13 partner countries (Algeria, Burkina Faso, Chad, Djibouti, Egypt, Ethiopia, the Gambia, Mauritania, Mali, Niger, Nigeria, Senegal and the Sudan) in their efforts to implement the GGWSSI. This support involves: (i) the development and validation of a harmonized regional strategy for effective implementation and resource mobilization of the GGWSSI; (ii) the preparation of detailed implementation plans and project portfolios in the 13 countries, identifying priorities and intervention areas and at least three cross-border projects; (iii) the development of a partnership and resource mobilization platform and a learning and networking platform for enhancing knowledge sharing, technology transfer and promotion of best practices across GGWSSI countries and among partners; (iv) the preparation of a capacity-building strategy and programme; and (v) the preparation of a communication strategy and action plan for engaging key target audiences and stakeholders in supporting implementation of the GGWSSI.

Among the priority interventions identified within the GGWSSI action plans developed to date is the restoration of forest landscapes and degraded lands in the GGWSSI priority intervention areas. Achieving this will depend on developing the capacity of the partners in the following areas:

- use of native species adapted to the local environmental, socioeconomic and cultural conditions;
- selection, production and use of a wide range of site-adapted planting material (genotypes) from native tree, shrub and

[4] Desertification refers to land degradation in arid, semi-arid and subhumid areas resulting from factors such as human pressure on fragile ecosystems, deforestation and climate change.

Box I2-1.
Acacia operation project – Support to food security, poverty alleviation and soil degradation control in the gums and resins producer countries

This project, developed and implemented between 2003 and 2010, was funded by Italian Cooperation. It aimed at strengthening the capacity of six pilot countries (Burkina Faso, Chad, Kenya, Niger, Senegal and the Sudan) to address food security and desertification problems through the improvement and restoration of the acacia-based agrosilvipastoral systems, and at sustainably developing the resins and gums sectors. The project benefited local communities engaged in harvesting and processing gums and resins. The project tested a microcatchment water-harvesting system (the Vallerani system)[1] and restored a total of 13 240 hectares. Local people were empowered though an intensive programme of capacity building on the use and application of the Vallerani system, nursery establishment and plant production, agricultural production, and harvesting and processing of gums and resins. Native tree species, including *Acacia senegal, Acacia seyal, Acacia nilotica, Acacia mellifera, Bauhinia rufescens and Ziziphus mauritiana*, were established by planting seedlings and by direct sowing. Herbaceous plants, such as *Cassia tora, Andropogon gayanus* and *Cymbopogon* sp., were established by direct sowing.

The project also focused on strengthening the Network for Natural Gums and Resins in Africa, which involves 15 member countries, through resource assessment, training programmes and information sharing.

The project published a working paper, "Guidelines on sustainable forest management in drylands in sub-Saharan Africa," in both English and French.

A regional meeting held in Addis Ababa, Ethiopia, on 3–4 March 2009 identified the need for a strategy to develop the outcomes of the pilot project into a programme large enough to address the magnitude of food insecurity, poverty, land degradation and desertification in the region, and to mitigate and adapt to climate change. The future programme must first focus on improving livelihoods through broadening the sources of income for local populations, while restoring degraded lands and increasing the productivity of agriculture, range and forest systems. These are cross-sectoral activities and the programme must adopt an integrated approach. The programme will have to be of sufficient scale to be seen as a major actor in regional initiatives, such as the GGWSSI. Such a programme would contribute to combating desertification, to the success of the GGWSSI and, above all, to improving the well-being of the whole population in the region.

Source: For further information, see http://www.fao.org/forestry/aridzone/62998/en/.

[1] See http://www.lk.iwmi.org/africa/West/projects/Adoption%20Technology/RainWaterHarvesting/26-ValleranisSystem.htm.

grass species, including production sufficient quantities of seeds and seedlings of adequate quality;
- application of the principles of forest landscape restoration planning to restore ecological integrity and enhance human well-being in the degraded forest landscapes and lands;
- promoting effective stakeholder participation and governance to ensure effective planning, design, implementation and sharing of benefits from afforestation and restoration;
- promotion of sustainable management of forests and rangelands to assist and enhance natural regeneration;
- promotion of multipurpose agrosilvipastoral systems and economically valuable native plant species to improve rural livelihoods;
- combined use of traditional knowledge and innovative forestation and restoration techniques, with particular focus on soil and

water conservation and management;
- promoting awareness of the contribution of forestation and drylands restoration to climate change adaptation and mitigation within the framework of carbon market schemes (e.g. Clean Development Mechanism, Reduced Emissions from Deforestation and Forest Degradation (REDD) and REDD+) and adaptation schemes;
- sustainable financing and investments (e.g. through payments for environmental services) and related policy issues;
- monitoring and evaluation of the performance of restoration initiatives, and the assessment of their long-term sustainability and economic and environmental impacts;
- considering restoration along the whole market chain value, from the seed to the final product.

To support the effective planning and implementation of restoration work in the priority GGWSSI areas, FAO launched a process for developing guidelines on dryland restoration based on a compilation of lessons learned from past and current forestation and restoration projects and programmes. As a first step, the Turkish Ministry of Forestry and Water Affairs, FAO, the Turkish International Cooperation and Coordination Agency (TIKA) and the German Agency for International Cooperation (GIZ) convened an international workshop in Konya, Turkey, in May 2012. This workshop, entitled "Building forest landscapes resilient to global changes in drylands

Box I2-2.
Support to the rehabilitation and extension of the Nouakchott green belt, Mauritania

This project was implemented between 2000 and 2007 by FAO and the Ministry of Environment and Sustainable Development of Mauritania, with financing from the Walloon Region of Belgium. The project objective was to foster conservation and development of agrosilvipastoral systems around Nouakchott, while at the same time combating encroachment of sand on the green belt around the city. The project engaged the local community and national authorities in planning and delivering activities and in selecting appropriate local plant and tree species. A total of 400 000 plants were grown in nurseries and used to fix 857 hectares of threatened land (inland and costal dunes).

The project employed both mechanical and biological fixation methods. Partners and beneficiaries were trained on field techniques and management of tree nurseries through a participatory approach involving the local community and the support and supervision of technical experts from the project. The project gave priority to the production and use of indigenous woody and grassy species. For example, *Aristida pungens* was planted on very mobile strip dunes in accumulation zones. Deflation zones were planted with *Leptadenia pyrotechnica, Aristida pungens* and *Panicum turgidum*, while other slow-growing woody species, such as *Acacia raddiana* and *A. senegal*, were planted in more stable intermediate zones. Local grassy species were sown using broadcast seed, while *Colocynthus vulgaris*, a cucurbit, was sown in pouches. Establishment rate depended on rainfall. Plantings on coastal dunes concentrated on halophytic species, including *Nitraria retusa, Tamarix aphylla* and *T. senegalensis*.

The techniques used and the lessons learned are presented in detail in an FAO forestry paper published in 2010, which is available in English, French and Arabic. The best practices identified are now being replicated in other regions of Mauritania and will be promoted for adaptation and implementation in Mauritania and other countries of the GGWSSI.

Source: FAO (Food and Agriculture Organization of the United Nations). 2010. *Fighting sand encroachment: lessons from Mauritania*, by C.J. Berte, with the collaboration of M. Ould Mohamed & M. Ould Saleck. FAO Forestry Paper 158. Rome.

– Analysis, evaluation and documentation of lessons learned from afforestation and forest restoration," aimed at:

- gathering lessons learned from past and ongoing forest restoration efforts in the countries involved in the GGWSSI;
- identifying key elements determining the success or failure of forest restoration projects and;
- discussing the comprehensive Forest Restoration Monitoring Tool, recently developed by FAO to guide planning, implementation and evaluation of field projects and programmes.

A number of successful forestation and forest restoration projects exist in the GGWSSI countries and these can be quickly upscaled to support the effective implementation of the initiative. These include the two projects implemented by FAO and its partners: the Acacia operation project – Support to food security, poverty alleviation and soil degradation control in the gums and resins producer countries (Box I2-1), implemented in six sub-Saharan African countries; and the Support to the rehabilitation and extension of the Nouakchott green belt, funded by the Walloon region (Belgium), and implemented in Mauritania (Box I2-2).

For more information on the Great Green Wall for the Sahara and Sahel Initiative, please visit www.fao.org/partnerships/great-green-wall

Insight 3

Invasive species and the inappropriate use of exotics

Philip Ivey

South African National Biodiversity Institute and Working for Water Programme

Sometimes the choice of plant used in restoration can have unexpected and dramatic consequences both at the site of restoration and beyond. This Insight highlights some examples in which plants introduced from elsewhere in the world to help restore disturbed environments resulted in invasion and great environmental damage.

Exotic or non-native trees, shrubs, creepers, succulents and grasses have all been used to rehabilitate sites after human or natural perturbation has removed indigenous vegetation cover. Many introductions of exotic plants happened late in the nineteenth century or early in the twentieth century, when understanding of the likely impacts of these species was limited and not considered.

- *Pueraria montana* (kudzu), indigenous to China, eastern India and Japan, was introduced in the United States of America as a forage and ornamental plant, but was also extensively used in soil stabilization and erosion control.[5] It is estimated that about 120 000 hectares had been planted with kudzu by 1946, and the species has since spread beyond the planted range. By 2004 it was reported to be present and invasive in 22 states of the southeastern United States, where it causes extensive damage by smothering indigenous vegetation. It is not surprising that this species has a local common name of "vine that ate the South."
- *Acacia cyclops* and *Acacia saligna* were both introduced in the 1830s into South Africa from Australia to stabilize dunes and protect roads from sand storms (Carruthers *et al.*, 2011) but they became invasive species in the Western Cape of South Africa. Successful implementation of biological control measures to reduce seed production of these species will reduce the long-term threats they pose.
- *Ailanthus altissima* (Tree of Heaven), native to China and northern Viet Nam, has been used for a wide variety of purposes, including erosion control, afforestation, shelterbelts and to line promenades in Europe and elsewhere in the world. Consequently, the species has established and become invasive in suitable, lower-altitude environments across all of Europe. For a comprehensive review, see Kowerik and Säumel (2007).
- *Carpobrotus edulis* is known by the descriptive local common name of "highway iceplant" in California. The common name refers to the species' extensive use as a landscape plant to secure disturbed environments along roads. Since its introduction it has spread into natural environments where it threatens natural vegetation in several different environments, from dune systems to scrublands. *Carpobrotus edulis* is also a significant problem in Mediterranean countries, particularly Portugal.

[5] http://www.na.fs.fed.us/fhp/invasive_plants.

It is important that we learn from the mistakes of the past and if possible do not repeat them. There are several international protocols in place to encourage better practices to reduce the likelihood of invasions. Article 8(h) of the Convention on Biological Diversity (CBD) calls on parties to "prevent the introduction of, control or eradicate those alien species which threaten ecosystems, habitats or species." The Aichi Biodiversity Targets[6] agreed under the CBD similarly address invasive species: "By 2020, invasive alien species and pathways are identified and prioritized, priority species are controlled or eradicated and measures are in place to manage pathways to prevent their introduction and establishment" (Aichi Target 9).

The International Standards for Phytosanitary Measures, prepared by the Secretariat of the International Plant Protection Convention (IPPC) deals with "environmental risks," including "invasive plants." The IPPC encourages each of its regions to set regional standards. In response to this, the European and Mediterranean Plant Protection Organization (EPPO) has set standards to provide support to members dealing with both quarantine pests and more recently invasive alien species, and members are encouraged to manage these through national phytosanitary regulations.

Hulme (2007) estimates that 80 percent of invasive alien plants in Europe were voluntarily introduced for ornamental purposes. In an effort to curb the influx of new invasive plant species to Europe, the EPPO, in collaboration with the Council of Europe, developed a *Code of conduct on horticulture and invasive alien plants* (Heywood and Brunel, 2011) aimed at the horticultural industry. To an extent the European code of conduct has been based around the St Louis Declaration of 2002,[7] which calls on horticulturalists and the nursery industry to ensure that unintended harm (risk of invasion) is kept to a minimum when new plant species are considered for introduction.

One of the key indicators used to assess whether a species is likely to be invasive in a particular environment is whether it has been invasive elsewhere in the world. There are numerous reference lists of invasive and weedy plant species, including Randall (2002), the *Invasive species compendium*[8] and the DAISIE (Delivering Alien Invasive Species Inventories for Europe) database.[9]

In order to achieve the targets set by the CBD and to reduce the likelihood of new invasive species being used by the horticultural industry for landscape rehabilitation, it is important that governments control imports of new plant species. Horticultural interests also should regulate their own businesses by adhering to the voluntary protocols to control invasive species. With adequate control and self-regulation, the errors of the past need not be repeated by environmental managers of today. With better knowledge of the risks posed by certain species, the goodwill of all stakeholders and much hard work, there is no reason why further potentially invasive species should be introduced for the purposes of environmental rehabilitation.

6 http://www.cbd.int/sp/targets/

7 http://www.fleppc.org/FNGA/St.Louis.htm

8 http://www.cabi.org/isc

9 http://www.europe-aliens.org/

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Part 2

THEORETICAL AND PRACTICAL ISSUES IN ECOSYSTEM RESTORATION

Part 2 presents issues that should be considered in all restoration efforts, irrespective of the local context and the specific methods used. Building on theoretical understanding of genetic processes, the authors discuss how selection, genetic drift and gene flow can affect outcomes of restoration efforts. Local forest remnants are widely considered to be ideal sources of propagation material because they are assumed to be well adapted to local conditions as a result of millennia of natural selection. However, it is often overlooked that the remnant forests may be too small to sustain viable populations, and may suffer from genetic drift that results in random loss of diversity (Chapter 2, Insight 4: Historical genetic contamination in pedunculate oak (*Quercus robur* L.) may favour adaptation and Chapter 4). Gene flow through pollen and seed dispersal can counteract negative implications of small populations (Chapter 5). Transferring genetic material over longer distances may, however, threaten indigenous genetic diversity and result in a loss of local adaptations (Chapter 6 and Chapter 3). However, such long distance transfers may be beneficial in certain circumstances (see Insight 4: Historical genetic contamination in pedunculate oak (*Quercus robur* L.) may favour adaptation). In most cases, little is known about the extent and distribution of genetic diversity of tree species used in restoration. Rules of thumb may exist for collecting and transferring propagation material in such cases, although those remain little studied in practice (Chapter 7).

The introduction to the theoretical concepts is followed by presentation of examples of their practical application and constraints faced in restoration efforts. Various types of propagation materials are discussed and guidance is provided on choosing suitable types for local contexts (Chapter 8). Considering the current proliferation of restoration efforts and the simultaneous degradation of natural tree populations of many species, little attention is usually given to the sustainable sourcing of massive amounts of propagation material (see Chapter 8 and Insight 6: Seed availability: a case study). Seed banks are effective and often-overlooked sources of material for those species that can easily be stored as seed (Insight 7: The role of seed banks in habitat restoration). Traditional ecological knowledge held by local and indigenous communities can be a valuable source of information on suitable tree propagation and management practices, not least because it has played an important role in shaping tree diversity for hundreds or thousands of years in many areas (Chapter 10). Finally, restoration efforts should not be planned in isolation but must carefully consider the local landscape context, recognizing and appreciating the needs and priorities of the various interest groups (Chapter 11).

Chapter 2

Seed provenance for restoration and management: conserving evolutionary potential and utility

Linda Broadhurst[1] *and David Boshier*[2,3]

[1] *CSIRO Plant Industry, Australia*
[2] *Department of Plant Sciences, University of Oxford, United Kingdom*
[3] *Bioversity International, Italy*

Diverse biological, cultural, environmental and socioeconomic conditions across the world demand diverse approaches to forest or habitat restoration and sustainable farming. Trees are a vital component of many farming systems, while a range of agroforestry systems have the potential to conserve native species as well as to diversify and improve the production and income of resource-poor farmers. Although native species are usually favoured in tree planting for forest or habitat restoration or by local people on farms, often only a limited range of management options and tree species (often exotics) are promoted.

Tree planting depends on a ready supply of germplasm (seeds or vegetative material) of the chosen species, which in turn requires consideration of what is the best or most appropriate source of seed. Inevitably the choice of seed source should be influenced by the objective of planting (e.g. for restoration or production, future adaptability or past adaptation) and the risks associated with particular seed sources (e.g. loss of adaptation, outbreeding depression, loss of diversity, genetic bottlenecks or contamination of native gene pools). Choice of seed source, both in terms of its location and its composition, can have important consequences for the immediate success and for the long-term viability of plantings.

Many tree species are outbreeding and generally carry a heavy genetic load of deleterious recessive alleles. This means that inbreeding, in particular selfing, can have negative impacts, including reduced seed set and survival resulting in poorer regeneration, progeny with slower growth rates and lower productivity, limited environmental tolerance and increased susceptibility to pests or diseases. Consequently, the use of genetically diverse germplasm is vital if plantings are to be productive, viable and resilient. Intraspecific genetic diversity may, however, be limited by several factors related to the sourcing of seed. For example, farmers, nursery managers and commercial collectors may collect seed from only a few trees as this requires less effort than collecting from many trees; however, this captures only a small amount of the variability present. In addition, variability in fertility between trees can contribute to a rapid accumulation of relatedness and inbreeding in subsequent generations. Genetic issues can also be of particular concern for nursery material, where inbred material may survive benign nursery conditions but be genetically compromised for survival and growth when planted out in the wider environment.

2.1. Local versus non-local seed

Many guidelines for sourcing seed to restore plant populations and communities advocate the use of local seed under the premise that this will be better adapted to local conditions and deliver superior outcomes through improved survival and growth (Broadhurst *et al.*, 2008 and references therein). Apart from the possibility of non-local seed being maladapted to local conditions, using seed collected close to a restoration site is also predicted to prevent negative outcomes, such as intraspecific hybridization (potentially) resulting in outbreeding depression, superior introduced genotypes becoming invasive and impacts on associated organisms such as bud burst occurring prior to herbivore emergence, and to help maintain a range of biotic interactions with pollinators and pathogens (Linhart and Grant, 1996; Jones, Hayes and Sackville Hamilton, 2001; Cunningham *et al.*, 2005; Vander Mijnsbrugge, Bischoff and Smith, 2010). Although the importance of local provenance in habitat conservation and restoration remains contentious (e.g. Sackville Hamilton, 2001; Wilkinson, 2001), the concept is easy to understand and the message is therefore attractive and easy to "sell" (see, for example, www.floralocale.org). Hence the *General guidelines for the sustainable management of forests in Europe* (MCPFE, 1993) state that "native species and local provenances should be preferred where appropriate." Forest certification and timber labelling standards also require action to conserve genetic diversity and to use local provenances (e.g. PEFC, 2010; UKWAS, 2007). Grants for tree planting often require the use of local material, although this may depend on the purpose of planting (e.g. Forestry Commission, 2003).

Despite such requirements to source seed locally, many guidelines provide little direction as to how this should be evaluated (Broadhurst *et al.*, 2008), with practitioners often interpreting guidelines in a spatial context at a range of scales (e.g. as small as a particular farm or wood to as large as a country). To fully evaluate the superiority of local seed requires complex experiments and long-term monitoring that go beyond early effects on germination and growth, and that are beyond the scope of most restoration projects. Consequently, there is often little empirical evidence for deciding how local a seed source should be. Should seed come from the same wood, the same watershed, the same county or the same country? Is geographical or ecological distance more important (e.g. Montalvo and Ellstrand, 2000)? With limited information about the extent and scale of adaptive variation in native trees, discussion about suitable seed sources often emphasizes "local" in a very narrow sense or within political boundaries, rather than being based on sound evidence of the scale over which adaptation occurs.

The requirement to use locally collected seed has been given such precedence that restoration projects have occasionally been abandoned because of a lack of appropriate local seed sources (Wilkinson, 2001). Use of native species in both restoration and on farms has also been limited by a lack of basic information on seed storage and germination and establishment methods; a reflection of the historical emphasis on plantation forestry with a limited range of exotic species.

2.2. Basic concepts and theory

It is worth considering some basic concepts to appreciate to what extent and at what scale local adaptation may apply. The forces of natural selection may vary in space, resulting in genotype × environment interactions for fitness. In the absence of other forces and constraints, such divergent selection should cause each local population to evolve traits that provide an advantage under its local environmental conditions (i.e. its habitat), regardless of the consequences of these traits for fitness in other habitats. What should result, in the absence of other forces and constraints, is a pattern in which genotypes of a population would have on average a higher relative fitness in their local habitat than genotypes from other habitats. This pattern and process leading to it is local adaptation (Williams, 1966).

However, local adaptation may be hindered by gene flow, confounded by genetic drift, opposed by natural selection as a result of temporal environmental variability and constrained by a lack of genetic variation or by the genetic architecture of underlying traits. Thus, although divergent natural selection is the driving force, these other forces, in particular gene flow, are integral aspects of the process of local adaptation. Owing to such forces, local adaptation is not a necessary outcome of evolution under spatially divergent selection (Kawecki and Ebert, 2004). Environmental heterogeneity also favours the evolution of adaptive phenotypic plasticity. Where there are no costs of and constraints on plasticity, a genotype that produces a locally optimal phenotype in each habitat would become fixed in all populations. Adaptive phenotypic plasticity would lead to adaptive phenotypic differentiation, but without underlying genetic differentiation. Lack of plasticity is thus a prerequisite for local adaptation.

In summary, factors predicted to promote local adaptation include: low gene flow (i.e. restricted pollen or seed dispersal, or strong habitat fidelity), strong selection against genotypes optimally adapted to other habitats but moderate selection against intermediate genotypes (most likely under moderate differences between habitats with respect to traits under selection), little temporal variation in the forces of selection, small differences between habitats in size and quality (e.g. the amount of resources) and costs of or constraints on adaptive plasticity.

2.3. Historical perspective of local adaptation

The extent to which observed morphological and growth differences in plants are under genetic control and related to the environment in which a population occurs, has formed fertile ground for research. Linnaeus reported as early as 1759 that yew trees brought to Scandinavia from France were less winter hardy than indigenous Swedish yews. In his classical research, Turesson (1922) studied populations of several herbaceous species in transplant common garden experiments, demonstrating the widespread occurrence of intraspecific, habitat-related genetic variation and introducing the term "genecology." Clausen, Keck and Hiesey (1940) extended study of the expression of population adaptation to environmental differences by using climatically different sites over a range of altitudes. Subsequent research has shown that such genetically related adaptive variation is widespread in herbaceous species with low levels of gene flow under strong selection pressures (see summary in Briggs and Walters, 1997). There are many key differences between herbaceous plants and trees, where long life cycles, wide distributions and extensive gene flow (pollen and seed dispersal) would tend to suggest more extensive scales and patterns of adaptation, with differences most likely to occur at the geographic and altitudinal extremes of species ranges.

2.4. The scale of local adaptation in trees: how local should a seed source be?

Evidence for strong local adaptation effects, especially in trees, remains mixed and such adaptation is very difficult to predict (Ennos, Worrell and Malcolm, 1998; Montalvo and Ellstrand, 2000; Joshi *et al.*, 2001; Hufford and Mazer, 2003; Bischoff *et al.*, 2006; Leimu and Fischer, 2008). Provenance and progeny field trials have shown that while genotype × environment interaction occurs in many tree species, this may not be expressed as a home-site advantage (i.e. provenance performance is unstable across sites, but not as a result of greater fitness of local seed source). Geographical proximity may be a poor indicator of adaptive fitness (e.g. *Betula* spp.; Blackburn and Brown, 1988) and also stability, with some provenances that show stable performance across sites originating from sites adjacent to unstable performers (e.g. Kleinschmit *et al.*, 1996).

The northern hemisphere forestry literature suggests that latitudinal or altitudinal gradients, or both, can be important for detecting the scale of local adaptation, but that other factors such as habitat, rainfall and topographical differences can also be significant (Ennos, Worrell and Malcolm, 1998). There is evidence for adaptive variation over reasonably short distances in a number of conifer tree species in western North America, owing to features such as aspect and altitude (e.g. Adams and Campbell, 1981; Sorensen, 1994). This is particularly marked in areas with oceanic climates, where environmental gradients are much steeper than in more continental sites. Field, greenhouse and laboratory studies on conifer species in the northwestern United States show that a significant proportion (typically 25–45 percent) of the genetic variation within populations is accounted for by climatic (e.g. rainfall and temperature) or location (e.g. latitude, altitude, slope aspect, distance from ocean) variables that reflect environmental factors specific to each location. There are often differences between provenances from warmer and colder climates, the former showing adaptation to the longer growing season in lower latitudes but suffering from early or late frosts when moved too far into higher latitudes. The degree of risk in transplanting across a species' distribution is correlated more with environmental changes than with the geographical distance moved (Adams and Campbell, 1981; see Insight 5). This suggests that habitat matching may be a more useful means of determining where seed should be sourced than would be an arbitrary distance from the site to be restored. Provenance trials of a number of tropical tree species show that most morphological genetic variation occurs within rather than between provenances. In most of the species studied, ranking reversals (adaptation) or significant genotype × environment interactions only occur with large environmental site differences (e.g. dry vs wet zones, alkaline vs acidic soils). Unfortunately, almost nothing is currently known about local adaptation in temperate southern hemisphere species.

Currently there are too few studies from too few regions of the world to allow for predictions regarding the scale and importance of local adaptation for the myriad of life-history traits and evolutionary histories of tree species that require restoration. For example, Leimu and Fischer (2008) used only 32 species in their local adaptation meta-analysis, none of which were tree species. There is also a need for reciprocal transplant experiments (RTEs), which test the fitness of "home" and "away" genotypes within the sites from which the genotypes originate (Primack and Kang, 1989) and can mimic natural regeneration by establishing seedlings in a forest at close spacings to encourage early competition and with minimal intervention (e.g. little or no weeding). Germplasm selected and tested in forestry trials or plantations for growth, form and other commercial criteria may be less suited to the more competitive environment of semi-natural forests and restoration.

The scale over which species show adaptation to their environment depends on the degree of habitat heterogeneity, in particular the specific habitat characteristics that affect a species, and the interaction with gene flow. Dispersal levels may be a useful high-level predictor of the importance of local adaptation, under the premise that species with long-range gene flow are less likely to generate strong local adaptation, whereas restricted gene flow is more likely to generate genotypes adapted to their local environment. Extensive gene flow in widely distributed tree species suggests that local adaptation over a small geographic scale is unlikely unless selection forces are very strong.

2.5. Are non-local seed sources ever appropriate?

In highly modified or degraded landscapes, using non-local seed may be entirely appropriate or indeed the only option for restoration. Mitigating the negative physical effects associated with vegetation removal, such as loss of topsoil, altered hydrological flows or increased nutrient loads, may require specific germplasm that is able to cope with these conditions. For example, saline scalds in southern Australia that developed following the removal of deep-rooted perennials are not generally amenable to restoration using local species, let alone local seed. In these cases, planting saline-tolerant varieties of other species may be the only option to prevent further degradation of valuable agricultural land. The loss of diversity at genes of major effect may also require sourcing of seed from non-local populations. For example, small populations of self-incompatible plants can be mate-limited if diversity in the incompatibility locus is low, requiring seed from beyond the local area to introduce new mating types. However, impacts on local species and communities that may arise from using non-local seed need to be considered carefully, preferably prior to restoration and using an appropriate risk management framework (Byrne, Stone and Millar, 2011). This should also include analysis of the risk of not undertaking restoration and allowing landscape degradation and biodiversity loss to continue.

2.6. Local seed sources may not produce restoration-quality seed

Habitat fragmentation remains a major threat to biodiversity worldwide through the loss of populations and consequent altered biotic and abiotic processes (Bakker and Berendse, 1999; Eriksson and Ehrlen, 2001; Hobbs and Yates, 2003; Lienert, 2004). Unfortunately, some regions of the world have now reached a tipping point, such that whole biomes may be in danger of collapse (Hoekstra *et al.*, 2005). The most immediate consequence of fragmentation for use of native species in restoration and farm systems is limitations to seed supply following the loss of individuals and populations. But several negative genetic and demographic effects associated with fragmentation can also have an impact on the quantity and quality of seed available. The removal of trees and populations from landscapes directly reduces genetic diversity, most of which is irreplaceable since genetic mutations accumulate slowly over long evolutionary periods (i.e. tens of thousands to millions of generations). Diversity is further eroded in small populations by drift resulting from random sampling within populations, as well as inbreeding as a result of trees in remnant populations often being more highly related than those in larger populations (Barrett and Kohn, 1991; Ellstrand and Elam, 1993). Reduced fitness and productivity are commonly documented effects associated with genetic erosion and inbreeding, both of which can have an impact on a population's ability to persist in stressful situations or changing environments (Frankham, Ballou and Briscoe, 2002; Hughes *et al.*, 2008). Other negative outcomes include poor reproductive success, smaller, poor-quality plants and increased susceptibility to pests and pathogens (Lienert, 2004 and references therein). Over time, this exposes small populations to decline through recruitment failure (Figure 2.1) and limits their utility as appropriate seed sources for restoration. Limited seed supply and poor-quality seed are two major impediments to the successful planting of native species and restoration of native vegetation, especially at the landscape level.

Worldwide analyses of fragmentation impacts on plant reproduction indicate that some species are shifting towards selfing (Aguilar *et al.*, 2006; Aguilar *et al.*, 2008; Eckert *et al.*, 2010), but how this translates to seed production depends largely on reproductive strategy. For example, species that cannot self or mate with close relatives (self-incompatible) will not produce seed unless pollinated by distantly- or non-related trees and small, self-incompatible populations are often characterized by reduced seed production, severely limiting quantities available for restoration. In contrast, species that can self and mate with close relatives (self-compatible) continue to produce seed but this is often less fit, being smaller, slower to germinate and with poorer survival (Buza, Young and Thrall, 2000; Young

Figure 2.1.
Simplified representation of how low genetic diversity and inbreeding can impact on plant population persistence and seed production in small populations of plants

et al., 2000; Mathiasen, Rovere and Premoli, 2007). Restoration using this seed is therefore likely to produce poorer results than expected and over the long term is less likely to develop into a self-sustaining population. A requirement that only local seed be used for restoration can drive practitioners to use seed from small, inbred populations that are unlikely to produce positive long-term restoration outcomes, but rather create more small, inbred populations, with limited long-term persistence. One consideration is that populations restored with a narrow genetic base may be limited in their ability to respond to the rapid predicted shifts in climatic variables (Helenurm, 1998).

2.7. Adaptation and climate change

There are theoretical reasons that underlie observed patterns of adaptive variation; these also suggest that many tree species over large areas may fail to show local adaptation at a very narrow scale. The prevalence of extensive gene flow may counteract selection, while the temporal variation in selective forces that trees experience (e.g. yearly variation in temperature, frosts or rainfall) is likely to have a stabilizing effect rather than the directional selection that would lead to highly localized adaptation. Given the long life of trees, the environment is also likely to have altered over the lifespan of a tree or only a few generations, such that a particular site no longer experiences the same conditions under which the trees originally evolved. These factors explain the relative lack of adaptation over short distances in many tree species. Temporal variation in environment is particularly important for trees, not only with respect to past adaptation but also in the context of predicted climate change (e.g. Broadmeadow, Ray and Samuel, 2005), and thus undue emphasis on local seed sources may also cause problems.

2.8. Benefits of using larger but more distant seed sources

Using large populations as primary seed sources for restoration not only ensures that seed quality will be higher but also that larger quantities are available. In many cases a population of 100–200 plants would be large enough to provide good-quality seed, but more than 400 plants may be needed for some species. A good restoration outcome is also more likely if the habitat of the site to be restored is matched as closely as possible with that of the nearest large population. Seed from these large populations could be augmented with that collected from small populations closer to the restoration site to capture any useful genetic diversity they may contain (Broadhurst *et al.*, 2008). To capture as much genetic diversity as possible from large populations, as many plants as practically possible should be sampled broadly across the site, collecting from a range of cohorts, from various sides of plant canopies without disrupting biotic associations that also rely on this seed. Breed *et al.* (2012) reviewed such strategies for sourcing restoration seed (Box 2.1) and summarized their suitability for mitigating climate change and habitat fragmentation impacts (Table 2.1).

The mixing of introduced and native germplasm raises the issue of outbreeding depression; the potential problem of reduced vigour as adapted gene complexes are broken up or the proportion of locally adapted alleles is reduced. As with local adaptation, evidence for outbreeding depression comes from herbaceous species that show highly localized adaptation (see Hufford and Mazer, 2003) and there is little evidence for its occurrence in trees at distances of less than hundreds of kilometres (e.g. Hardner *et al.*, 1998, Boshier and Billingham, 2000). For example large-scale importation of cheap seed from Eastern Europe has shown problems of maladaptation in Britain. But it seems unlikely that use of material from maritime France that matches future climate predictions (Broadmeadow *et al.*, 2005) and of similar phylogeographic origins will face such problems, nor lead to outbreeding depression problems on introgression with British material.

2.9. Conclusions

Any genetic conservation policy for native trees should aim at conserving the evolutionary potential of their populations, rather than at preserving a particular genetic structure and status. The extent and scale of local adaptation in many tree populations, and thus its practical importance to restoration efforts, remain in doubt. While there is a need for more field trials, both of the traditional provenance or progeny and RTE types, to provide more information on the scale of adaptation, planting of native trees continues apace and demand for seed from certified sources increases. There is good evidence to suggest that emphasis on a very restricted view of what is "local" will not lead to better-adapted tree populations and is more likely to lead to use of stock of limited genetic diversity than would a broader approach.

It has been argued that, given the lack of extensive trials investigating adaptive variation in native tree populations, the precautionary principle should be adopted in sourcing germplasm for planting trees (e.g. Flora Locale, 1999; UKWAS, 2007). This is expressed as the use of local seed, although the subsequent view of what constitutes the local population varies from a particular forest to large seed zones. However, given current evidence for trees, i.e. clear dangers from inbreeding and loss of genetic diversity, with extensive gene flow and adaptation at a broad scale, it seems more logical to apply the precautionary principle in terms of ensuring the use of genetically diverse material with the capacity to adapt to current and future conditions.

Box 2.1.
Summary of alternative strategies for sourcing seed for restoration

Strict local provenancing: collecting seeds from plants that are located physically very close to the revegetation site (e.g. Natural England, United Kingdom: 5 miles; Western Australian Forest Management Plan 2004–2014: 15 km).

Relaxed local provenancing: collecting seeds with a bias towards certain ecological criteria, and avoiding small population fragments (e.g. Australian FloraBank: soil type, altitude and climate).

Predictive provenancing (Sgrò, Lowe and Hoffmann, 2011)**:** use of naturally occurring genotypes experimentally determined to be adapted to projected conditions. This technique requires data on local adaptation of target species (e.g. by reciprocal transplant experiments), as well as climate projections for these species at a revegetation site (e.g. by bioclimatic modelling).

Composite provenancing (Broadhurst *et al.*, 2008)**:** collecting a mixture of seed that attempts to mimic natural gene-flow dynamics. For example, recommended proportions of seed collected from local, intermediate and distant distance-classes could be determined by estimating the pollen dispersal kernel for target species.

Admixture provenancing (Breed *et al.*, 2012): collecting seed only from large populations, focusing on capturing a wide selection of genotypes from a diversity of environments with no spatial bias towards the revegetation site. These seeds are then admixed for sowing or planting, generating a population with a mixture of genotypes from a wide array of provenances.

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TABLE 2.1.
Suitability of provenancing techniques under climate change with habitat fragmentation

Provenancing technique	Adaptive potential benefits	Genetic rescue benefits	Low genetic load	Suitable with high uncertainty	Economically efficient	Likely population success
Strict local					x*	
Relaxed local					x*	
Predictive		x	x			x†
Composite	x	x			x	x
Admixture	x	x		x	x	x

* May experience high failure rates, negating the economic benefit.
† Benefit rests on successfully matching genotype fitness with future conditions.

Source: Breed *et al.* (2012).

Current threats to the maintenance of genetic diversity come principally from poor practice in seed collection; undue emphasis on restricting the area of collection or poor instruction of collectors can limit the number of trees and hence genetic diversity sampled, leading to the establishment of trees with restricted genetic diversity and limited future adaptive potential. A study of the few remnant ash and rowan trees in the denuded Carrifran valley in southern Scotland showed that large amounts of genetic diversity are maintained, making them suitable for use in restoration despite their highly fragmented nature (Bacles, Lowe and Ennos, 2004). In contrast, some of the locally sourced material planted as part of the Carrifran wildwood restoration project was shown to be low in genetic diversity (Kettle, 2001), presumably because of poor collection practices, which impose limitations on the future potential of the population.

It is disturbing to contemplate that some of the poorest seed sources exist in the very regions where restoration is most needed and that continued requirements for using local seed simply perpetuate the problem. In many regions of the world with fragmented forest populations, being able to reliably source large volumes of quality seed of native species can be challenging. Not only are there fewer populations from which seed can be collected, but fragmentation has split continuous populations into much smaller and more isolated remnants, which can impact the quality and quantity of seed (e.g. Lowe *et al.*, 2005). The implications from this are that (i) remnant vegetation contains all of the diversity that is left that is extremely valuable, and (ii) it is important that most of the diversity that does remain is used for restoration (i.e. avoid over-collection from a few populations). In regions where fragmentation is high, should the rules for using local seed change? Can we afford the luxury of being too restrictive about seed sources? Are small, fragmented and probably inbred populations so precious that we cannot source seed from beyond our comfort zone?

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Chapter 3

Continuity of local genetic diversity as an alternative to importing foreign provenances

Kristine Vander Mijnsbrugge[1,2]

[1] *Research Institute for Nature and Forest, Belgium*
[2] *Agency for Nature and Forest, Belgium*

A population of trees or shrubs is autochthonous if it has regenerated naturally since its arrival after the last glaciation; any human intervention in breeding should have occurred with strictly local material only. For long-lived species such as trees, autochthony assumes a continuous presence at a given site since post-glacial immigration (Kleinschmit, Kownatzki and Gegorius, 2004). This implies a continuity of local genetic diversity after thousands of years of natural selection. Trees and shrubs that belong to native species but are imported from other climatic zones or geographic regions are not autochthonous.

After many years of neglect, the use of native species in afforestation and landscape programmes is gaining importance all over the world, based on the basic underlying ecological principle that native species and genotypes will be well adapted to local conditions and will have co-evolved with other components of local forest ecosystems. This has led to massive plantations of indigenous tree and shrub species in Western Europe, not only in forestry but also for native woodland restoration and other landscape plantings, such as thickets, wooded banks and hedgerows. A major challenge is to ensure that planting material used represents the genetic variation and diversity within native species. Several initiatives have been developed in various European countries to promote the use of locally sourced seeds for the production of planting stock (e.g. Belgium: Vander Mijnsbrugge, Cox and Van Slycken, 2005; Germany: Kleinschmit, Leinemann and Hosius, 2008; Denmark: Kjaer *et al.*, 2009). Here we describe in detail the programme on the production of autochthonous planting stock in Flanders, Belgium.

3.1. Why should autochthonous diversity be protected?

There is a high demand for "native" planting stock in Flanders, Belgium, and to a broader extent in many Western European countries. The use of native planting material is promoted by a wide range of public organizations. However, planting stock of native material in commercial nurseries is largely not autochthonous. Seeds of native species are often imported, originating from foreign provenances, often in Eastern European countries. This is especially true for shrub species. For trees, in the European Union, Council Directive 1999/105/EC of 22 December 1999 (Council of the European Union, 2000) regulates the marketing and transport of forest reproductive material through an obligatory certification system indicating the origin of the material (although control in practice is not perfect). However, certification is not obligatory for shrubs, and shrub germplasm is commonly imported from Eastern and Southern Europe where cheaper seed is available. Nursery managers often do not know or are not interested in the exact origin of the seed they obtain. Tree seed may also be imported when seed is not available from officially approved sources or supplies are too limited to meet requirements.

Introduction of non-local material can have numerous negative consequences. Non-autochthonous planting stock may be poorly adapted to local growing conditions, which can lead to negative consequences such as lower fitness (e.g. McKay *et al.*, 2005; Krauss and He, 2006; Edmands, 2007; Laikre *et al.*, 2010; Vander Mijnsbrugge, Bischoff and Smith, 2010). Problems may only become evident many years after seemingly successful establishment. Intraspecific hybridization of local and introduced genotypes may result in outbreeding depression, i.e. reduced fitness in subsequent generations, loss of genetic diversity and loss of adaptation, and less adapted characteristics can introgress into the autochthonous populations. The introduction of non-local material may also have negative effects on associated plant and animal species. Imported hawthorn (*Crataegus monogyna*) has been shown to flower several weeks earlier than native hawthorn, potentially threatening the insects and birds whose reproductive cycles are synchronized with this event (Hubert and Cottrell, 2007). In addition, purity of the species can be problematic in commercial planting stock. A genetic study on commercially available hawthorn in Flanders, grown from seeds imported from Hungary, showed that it comprised a mixture of *C. monogyna* and *C. monogyna* × *C. rhipidophylla* (Debeer, 2006).

3.2. Inventory of autochthonous woody plants

A simple way to ensure the continuity of local genetic diversity is the production of autochthonous planting stock. For this an overview is needed of the remnant autochthonous populations still present. A survey was conducted to locate remaining autochthonous populations in Flanders, Belgium, from 1997 to 2008. The evaluation of autochthony in the field was conducted following the methodology presented by Maes (1993). In short, areas of woody vegetation that are indicated as forest on historical maps are identified. Information on flora, soil conditions and geomorphology further refine the selection of potentially relevant sites. In the field, the woody vegetation is evaluated according to a set of criteria. The tree or shrub must be a wild variety and old. No evidence must be seen of plantation (e.g. trees in lines). The site must be located within the natural geographic range of the species, and the growth conditions correspond to the ecological requirements of the species. The tree or shrub must be present on similar sites in the surrounding area. A variety of plants in the tree, shrub or herb layer is indicative of undisturbed woodland and ancient forests. If hedges or wooded banks have been planted with locally sourced material the plants can be considered autochthonous.

The findings show that autochthonous woody plants have become seriously endangered in Flanders, with only about 6 percent of the current forest cover holding autochthonous woody plants. Several causes for this loss of autochthonous material are evident. Only 11 percent of Flanders is now forested and what there is is highly fragmented as a result of centuries of intensive forest use. Small fields have been replaced by large, open expanses of farmland, with the consequent disappearance of wooded banks, old hedges and small forests on farmland.

The inventory data (in Flemish) are accessible on the internet (www.natuurenbos.be).

3.3. Producing autochthonous planting stock

The Agency for Nature and Forest (ANB), under the Flemish Forest Administration, has been collecting seed from inventoried sites since 1998 to produce autochthonous planting stock. Seed is collected from natural populations present on inventoried sites (so-called *in situ* collecting) following general guidelines for appropriate collection methods. Sites adjacent to plantations of the same species are omitted because of the risk of cross-pollination from unknown provenances. Seed is collected from at least 30 seed-bearing plants per species within each region of provenance. Region of provenance, a term commonly used in forestry, is an area within which movement of plant material will not negatively affect the fitness of the populations in the long run. In Flanders, the surveyed sites are mostly fragmented, small and are not managed for seed production. Therefore, several sites must be visited to find 30 seed-bearing trees or shrubs for every species. This implies a time-consuming and costly effort. The Flemish legislation (Anonymous, 2003a), which follows Council Directive 1999/105/EC, allows mixing seed lots within a region of provenance. This practice guarantees a good genetic variability in the derived planting stock. A genetic study on sloe (*Prunus spinosa*) in Flanders showed that old autochthonous hedges dominated by sloe may show low within-population genetic diversity. In this case, mixing of seed lots from different autochthonous locations is specifically advised (Vander Mijnsbrugge *et al.*, in press.

Until now, the planting stock has been grown in two government nurseries located in Koekelare and in Brasschaat. However, the decision has been taken to close them, mainly for financial reasons. Future planting stock will be grown increasingly in private nurseries under contract. The autochthonous planting stock is used only in forests owned by or managed by ANB. As seed collection, growth and planting are all performed within the forest administrative boundaries, no certification or control system is involved.

Since 1998 seeds have been collected also by public organizations called Regional Landscapes ("Regionale Landschappen") that are working to protect and enhance the local authenticity of rural landscapes (Table 3.1). Here, all planting stock is grown in private nurseries under a sales contract. The seeds and derived planting stock are not certified, and the work of the nursery is not controlled by any official agency, necessitating a relationship of trust between the client and the nursery. Again, this autochthonous planting stock is used in the Regional Landscapes' own projects, mainly landscape plantings such as hedgerows, wooded banks, on farms, etc., and can also be sold to local people.

Since 2004 seeds can be collected on inventoried sites that are officially approved as a seed source, primarily under the category "source identified" (as defined by the Council Directive on the marketing of forest reproductive material). At least 30 seed-bearing trees or shrubs of the same species must be present on such sites, with a good score for autochthony. There must be no non-autochthonous plantations in the vicinity. Autochthonous stands showing traits of silvicultural value are approved under the category "selected." Five stands of *Alnus glutinosa* have been given this designation. Private nurseries can collect seeds from these officially approved seed sources and obtain a certificate from an independent governmental control agency that proves the origin of the seeds. The landowner of the collection site can charge those wanting to collect seeds, although in general private nurseries are not willing to pay large sums. Major problems faced by certified *in situ* collections are the reluctance of landowners to agree to the designation of a woody population on their property as an official seed source, the laborious process of approval of the sites, the small number of sites that meet the requirements for approval, and lack of management for high seed production. A major advantage of the system is that certified planting stock becomes available to a broader public.

3.4. Seed orchards

Seed orchards hold many advantages over *in situ* collecting. They produce large amounts of seed and at the same time preserve the gene pool of

TABLE 3.1.
Seeds and berries collected between 2006 and 2010 by Regional Landscapes (public organizations) from autochthonous populations in Flanders

Species	Fresh weight (kg)*				
	2006	2007	2008	2009	2010
Acer campestre	70.5	60.0	70.5	66.3	56.7
Alnus glutinosa	35.5	64.3	41.4	28.0	61.2
Carpinus betulus	116.3	159.3	11.5	141.4	88.0
Cornus sanguineum	26.0	39.8	26.0	11.3	6.4
Corylus avellana	10.2	229.3	26.5	56.7	71.9
Crataegus laevigata	4.7	11.6	4.0	2.8	0.9
Crataegus monogyna	111.8	44.1	59.8	107.6	103.8
Crataegus spp.	397.1	465.1	398.9	458.5	309.6
Euonymus europaeus	10.1	11.2	28.6	24.6	21.7
Fraxinus excelsior	105.3	2.5	62.4	5.6	226.4
Ilex aquifolium	18.1	5.0	5.5	6.0	5.9
Malus sylvestris	–	1.8	–	–	0.3
Mespilus germanica	1.5	42.8	5.1	2.5	9.4
Prunus padus	2.4	0.1	0.7	7.0	6.0
Prunus spinosa	218.3	121.1	24.8	113.7	107.9
Quercus robur	608.0	422.0	33.3	390.4	460.9
Rhamnus cathartica	1.0	5.4	4.4	–	–
Rhamnus frangula	13.7	20.4	34.0	51.9	56.0
Rosa arvensis	0.3	1.0	1.8	1.0	1.1
Rosa canina	26.2	16.5	35.7	11.3	19.3
Rosa corymbifera	–	–	7.8	–	–
Rosa spp.	1.0	1.4	–	5.0	5.4
Sambucus nigra	–	–	–	5.0	2.5
Sorbus aucuparia	109.5	67.0	39.1	135.4	66.9
Tilia cordata	–	–	–	1.5	–
Viburnum opulus	110.1	110.3	61.6	48.2	72.5
Total	**1997.5**	**1901.9**	**983.1**	**1681.4**	**1760.7**

* Uncleaned fresh weight.

the autochthonous populations from which the plants in the orchard originate. A programme initiated by the Research Institute for Nature and Forest and ANB for the creation of autochthonous seed orchards started in Flanders in 1999. Seed orchards have been established for all woody species that are regularly or occasionally planted. Basic material for these is collected at the inventoried sites. The objective is to represent the genetic diversity of the autochthonous populations present in a region of provenance. There are four main regions of provenance in Flanders, with an average area of 3000 km^2. Thus, theoretically, there should be four seed orchards for every woody species for which planting stock is desirable, one for each region of provenance. In practice, the number of orchards established differs for various reasons, such as the natural distribution pattern. For example, the nutrient poor soils in the north of Flanders (regions of provenance "Kempen" [KEM] and "Vlaamse Zandstreek" [VZA]) are characterized by a spectrum of species that differs from that found on the more nutrient-rich soils in the south (regions of provenance "Brabants District Oost" [BDO] and "Brabants District West" [BDW]). Thus, for example, seed orchards for *Eonymus europaeus*, a species found on nutrient rich soil types, have been established for only the BDO and BDW regions. Few relict populations remain for some rare and dispersed species such as *Tilia cordata*, *Ulmus laevis* or *Malus sylvestris*, and as a result orchards have been created using basic material from the whole of Flanders. Similarly, orchards for the whole of Flanders have been established for seemingly abundant species but for which autochthonous populations are rare, such as *Quercus petraea* or *Populus tremula*.

The most clearly authenticated autochthonous trees and shrubs are propagated, mainly vegetatively, from geographically scattered sites within the region of provenance. The use of vegetatively propagated plants ensures that they are genetically identical to the parent tree and ensures there is no pollution from non-autochthonous sources. In evolutionary terms, only one generation of exchange of genetic information is missed. The disadvantage is that vegetative propagation is difficult and expensive, particularly for recalcitrant genera such as *Quercus*. Experienced greenhouse technicians are indispensable. Labour- and cost-intensive *in vitro* techniques are not used. For trees with economic importance, the orchard clones can serve as parent material for breeding in future. Every seed orchard contains a minimum of 50 genotypes per species, collected from at least five different sites, and up to four ramets per genotype. An ideal seed orchard contains 200 plants. In addition, the aim is to duplicate each seed orchard at another location within the region of provenance.

Once established, the autochthonous seed orchards are officially approved as seed sources (category "source identified") and the seeds from them can be certified. The first plantations date from 2003 and planting is ongoing. The majority of orchards are situated on land owned and managed by ANB, while some have been established on municipal land and land owned by nature conservation organizations. By October 2011 a total of 14 339 plants had been planted in 90 seed orchards at 25 different locations in Flanders (Table 3.2). Shrub species in several orchards are fruiting and certified seeds are being collected by private nurseries and a commercial seed merchant (there is only one in Flanders). A major problem facing the nurseries is the technical and administrative inefficiency of a large number of small regions of provenance; other European countries have fewer, larger regions of provenance. Small countries tend to define small regions of provenance, mainly because of the absence of a pan-European consensus on the proper way to delineate them. The geographic scale of local adaptation is difficult and time consuming to measure for long-lived perennials.

3.5. Promotion of use

Flanders has a state-funded system for subsidizing (re)forestation that promotes the use

TABLE 3.2.
Number of individuals in the autochthonous orchards in Flanders by region of provenance (October 2011)

Species	BDO	BDW	VZA	KEM	Flanders	Total
Acer campestre	–	198	–	–	–	198
Carpinus betulus	–	–	157	–	303	460
Cornus sanguineum	–	509	–	–	–	509
Corylus avellana	73	299	205	225	–	802
Crataegus monogyna	572	759	629	–	–	1960
Euonymus europaeus	268	362	–	–	–	630
Fraxinus excelsior	177	100	161	108	–	546
Juniperus communis	–	–	–	742	–	742
Malus sylvestris	–	–	–	–	199	199
Mespilus germanica	–	216	218	–	–	434
Populus tremula	–	–	–	–	96	96
Prunus avium	–	–	–	–	181	181
Prunus insititia	–	–	–	–	20	20
Prunus padus	289	467	–	498	–	1254
Prunus spinosa	–	363	37	–	–	400
Quercus petraea	–	–	–	194	–	194
Quercus robur	–	–	–	117	–	117
Rhamnus frangula	–	183	238	547	–	968
Rosa arvensis	–	131	–	–	–	131
Rosa canina	–	203	243	–	–	446
Salix alba	76	–	126	–	–	202
Sorbus aucuparia	–	315	175	442	–	932
Tilia cordata	–	–	–	–	346	346
Tilia platyphyllos	–	–	–	–	142	142
Ulmus laevis	–	–	–	–	482	482
Viburnum opulus	541	540	405	462	–	1948
Total	**1996**	**4645**	**2594**	**3335**	**1769**	**14 339**

*BDO: Brabants District Oost; BDW: Brabants District West; VZA: Vlaamse Zandstreek; KEM: Kempen.

of autochthonous provenances (Anonymous, 2003b). A basic subsidy supports the use of native species, the amount of subsidy depending on the choice of species (e.g. indigenous oaks receive the highest subsidy). An additional subsidy, with a fixed financial value, is granted for the use of specific autochthonous provenances that are indicated on the list of endorsed provenances, which lists all officially approved autochthonous seed sources and orchards. The list (in Flemish) is accessible on the internet (www.inbo.be). A major drawback is that subsidies are only for (re)forestation, not any other landscape plantations such as hedges or tree rows or wooded banks.

3.6. Discussion

The Flemish government has invested heavily in production of autochthonous planting stock, starting with a laborious inventory, followed by both *in situ* collection of seeds and the establishment of seed orchards for many native species, both trees and shrubs. As a rough estimate, over recent years about 1 million autochthonous plants have been grown annually in both government and private nurseries. The majority of the plants are from seed collected *in situ* and grown under sales contracts. However, officially approved seed orchards are now starting to produce seed and certified seed is becoming increasingly available for all interested forest nurseries.

The programme now faces two issues. The first concerns communication. When private owners or public organizations buy planting stock their decisions are influenced by price, and autochthonous stock is more expensive (albeit sometimes only slightly) than non-autochthonous planting stock. As a result there is a tendency to purchase non-autochthonous planting stock. Targeted communication is needed to make all stakeholders aware of the value of autochthonous provenances, and the importance of the continuity of local genetic diversity of autochthonous populations. A major challenge lies in providing a clear explanation of the role and importance of genetic diversity. People readily understand that low genetic diversity leads to fitness problems related to inbreeding, but do not realize that bringing differentiated populations can have negative consequences that may result in diminishing genetic diversity and fitness.

The second major issue is control. Private nurseries play a pivotal role in production of autochthonous stock. However, their primary purpose is to make a profit. Inevitably, some nursery managers may be tempted to increase their profits by selling non-autochthonous stock as (more expensive) autochthonous stock. Although genetic studies can distinguish autochthonous from non-autochthonous material, they require highly skilled staff and are too expensive and time-consuming to use as a general control mechanism. Thus, controls during seed collection and growth in the nursery are the major (general) tools at hand.

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Insight 4

Historical genetic contamination in pedunculate oak (*Quercus robur* L.) may favour adaptation

Sandor Bordacs

Central Agricultural Office, Department of Forest and Biomass Reproductive Material, Hungary

The pedunculate oak (*Quercus robur* L.) in Central Europe was intensively managed in the past. Basically, the oak stands have been artificially reforested, using intercropping practices. Large numbers of acorns were planted, and some of these were imported from distant populations to improve the quality of oak wood expected.

The Slavonian oak, a local provenance of pedunculate oak, is reported to have a distinct population, mostly located in the Slavonian Plain in Croatia, southeastern Europe (Mátyás, 1972; Klepac, 1981). This area was largely unaffected by humans for almost 400 years from the fifteenth century because of frequent wars and military actions. Forests started to be harvested in the late nineteenth century. The stands were composed of 200–300-year-old huge oaks (up to 40 metres tall and yielding 40–50 m^3 of wood) with excellent wood quality. The largest tree on record had a breast-height diameter of 260 cm and yielded 64 m^3 of building timber and was sent to the Exposition Universelle in Paris in 1900. As a result of these qualities, this provenance was in high demand in Europe.

Historical documents show that huge numbers of acorns were harvested all over Slavonia and taken to distribution centres in the former Austrian-Hungarian Empire. These centres distributed the acorns throughout Hungary (Kolossváry, 1975) and to other European states. Excellent growth in many stands has been reported since the 1880s in Austria, Czech Republic, France, Germany, Hungary and many other parts of Europe (Koloszár, 1982; Sabadi, 2003).

A survey of chloroplast DNA diversity in Europe (Petit *et al.,* 2002) has shown that the specific haplotypes of the Balkan strain are most common in the Slavonian oak stands, and many planted stands elsewhere in Europe have varying proportions of these haplotypes (Gailing *et al.,* 2007) which are indicative of Slavonian origin. The Slavonian oak stands have not only been acclima-

Figure I4-1.
A 112-year-old Slavonian oak stand

tized to local conditions but also entered local oak gene pools across Europe. For example, Slavonian oak genes for early and late bud burst have been identified provenances in Germany (Gailing *et al.*, 2007). This kind of genetic contamination might be beneficial, especially in Central Europe where most plantations have been established and where intensive climate warming is predicted in the next 50 years. "Imported" southern genes for traits such as late bud burst and drought and heat tolerance may help local oak populations adapt to the changing climate.

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Insight 5

The development of forest tree seed zones in the Pacific Northwest of the United States

Brad St Clair

United States Department of Agriculture Forest Service, Pacific Northwest Research Station, United States

Seed zones and seed movement guidelines contribute to the restoration of native ecosystems by ensuring adapted and resilient plant populations. Seed zones have a long history in the Pacific Northwest of the United States. Plantation forestry was initiated in the early twentieth century with the establishment of the Wind River Nursery by the United States Forest Service in 1910. The nursery was established in southwestern Washington State to reforest and restore large areas of bare land and understocked forests resulting from large forest fires and logging. Initially, foresters did not pay particular attention to the source of forest tree seed. Seed came from easily accessible locations, often lower elevation forests near population centres and at logging operations. By the 1930s, however, it was becoming evident that not all plantations were as productive as they could be, particularly those at higher elevations, when compared with adjacent naturally regenerated stands. The gradual decline of trees from non-local sources was also evident in two pioneering research studies begun in 1912: the Wind River Arboretum, which tested trees from around the world for their suitability to the Pacific Northwest, and the Douglas-Fir Heredity Study, which addressed questions of type and location of Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco) parents from which to collect seed. Mortality and poor growth of trees from off-site sources increased as stands aged, with a particularly sharp increase in the years after an extreme cold-weather event.

These observations led in the 1940s to the establishment of the first seed collection zones and seed collection guidelines for Douglas-fir. A system to certify the stand origin of forest tree seed was established by the mid-1960s, and in 1966 seed zone maps for Washington and Oregon were published. These maps were widely used and have served their purpose of ensuring adapted planting stock for reforestation and restoration. In the meantime, researchers have learned much more about geographic patterns of genetic variation in adaptive traits for a variety of forest tree species, primarily from short-term genecological studies such as those by Campbell (1986) and by Sorensen (1992). Genecological studies consider genetic variation as found in common garden trials, and relate that variation to the climates or physiography of seed sources. Consistent, sensible correlations between genetic variation and seed-source environments indicate that a trait has responded to natural selection and may be of adaptive importance. Based on results from genecological studies, seed zones in Washington and Oregon were revised, primarily enlarging them in latitudinal directions, but mostly maintaining elevation limits (Randall, 1996; Randall and Berrang, 2002). This is because forest trees in temperate and boreal regions are primarily adapted to minimum winter temperatures, which are largely associated with elevation. Adaptation to aridity is also important in some regions, and may be particularly important in the tropics. Species may also differ in the scale and patterns of genetic variation and adaptation, and the revised seed zones in Oregon and Washington take these differences into account. Some species, such as Douglas-fir, are tightly adapted to their environments and may be considered specialist species. Other species, like western red cedar (*Thuja plicata* Donn ex D. Don), are more generally adapted and may be considered generalist species. Consequently, Douglas-fir has many seed zones with relatively narrow elevation bands of 150 m, whereas western red cedar has fewer seed zones with elevation bands of 450 to 600 m (Figure I5-1). An additional benefit of seed zones is that they contribute to maintaining genetic diversity and structure of forest trees at landscape scales that are likely most important for adaptation.

Seed zones and seed-movement guidelines have helped to ensure productive, healthy and diverse forests in the Pacific Northwest for the past half-century and more. What have we learned?

First, even with limited or no knowledge of genetic structure of a species, a reasonable assumption is that native populations are at least approximately adapted to their local environments (Savolainen, Pyhäjärvi and Knürr, 2007). The question then becomes, how local is local? Start somewhere. Make assumptions about the climatic or other environmental variables that are most important for adaptation, and delineate seed zones based on those assumptions. In the Pacific Northwest, initial seed zones were based on climatic variables of cold and drought,

Figure I5-1.
Seed zones for Douglas-fir and western red cedar in Oregon and Washington State, United States

for Douglas-fir

western red cedar

Source: adapted from Randall (1996); Randall and Berrang (2002).

vegetation types and physiography, especially elevation. Genecology studies indicated that many of those zones were too conservative, particularly in the north–south direction and particularly for some species that were later determined to be generalists. Revised seed zones took into account this new knowledge, but in the meantime, original seed zones based primarily on climate served their purpose.

Second, short-term genecology studies are valuable for indicating genetic structure important for adaptation and for delineating seed zones and seed-movement guidelines. These may be followed by longer-term reciprocal transplant studies or provenance tests to evaluate long-term adaptive responses, including estimating productivity given climates at the locations of seed sources and planting sites (see Wang, O'Neill and Aitken, 2010). An important finding from these studies is that each species must be considered individually, and that the patterns and scale of adaptation are not always obvious beforehand.

Finally, during the last few decades, scientists and land managers have recognized that climates are changing and have begun to consider management responses. Knowledge of genetic variation in adaptive traits is important for understanding responses of native populations to climate change and for evaluating management options to adapt to climate change, including planting populations adapted to future climates and ensuring genetic diversity for future evolution. The primary lesson from the development of seed zones in the Pacific Northwest is that, rather than waiting for the genetic knowledge to accumulate, it is better to act based on the best available knowledge, which may be from other species in other regions, and then to adjust management responses based on new knowledge from genetic studies.

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Chapter 4

Fragmentation, landscape functionalities and connectivity

***Tonya Lander*[1] *and David Boshier*[2,3]**

[1] *Natural History Museum, London, United Kingdom*
[2] *Department of Plant Sciences, University of Oxford, United Kingdom*
[3] *Bioversity International, Italy*

The needs of a large and growing human population have led to very high levels of habitat destruction. In more than half the world's biomes 20–50 percent of land area has been converted to human uses. The majority of conversion is for agriculture, which is expanding in 70 percent of countries, declining in 25 percent, and static in 5 percent (FAO, 2003). Although biogeographic regions differ markedly in the extent of habitat conversion to agriculture (Klein Goldewijk, 2001), in all regions at least 25 percent of the area had been converted to other land-uses by 1950. Tropical dry forests are the biome most affected, with almost half replaced by agriculture (Mace *et al*., 2005). Nearly 25 percent of tropical rain forest has been fragmented or entirely cleared (Mace *et al*., 2005), while temperate broadleaf and Mediterranean forests have experienced 35 percent or more conversion. However, global assessments show a decline in the rate of forest loss from 1990 to 2005 (Chazdon, 2008).

A major issue in land-use change is habitat fragmentation, defined as the reduction in area of a specific habitat type and division of the remaining habitat into smaller and spatially separated habitat patches as a result of replacement by anthropogenic land-uses, such as agriculture, human settlements or plantation forestry. The degree of fragmentation also varies between regions, with forest biomes in Africa and Europe twice as likely to be classified as "fragmented forest" compared with North and South America (Wade *et al*., 2003). Habitat fragmentation generally results in a complex landscape mosaic of native and human-dominated habitat types, which may have serious consequences for many species. This paper examines the impacts of fragmentation on the genetic viability of tree populations and how habitat connectivity and landscape functionality relate to the conservation and use of native tree species.

4.1. Genetic problems related to fragmentation

Maintenance of genetic diversity in trees is vital for the continued fitness, resilience, adaptation and evolution of their populations (Ellstrand, 1992; Garner, Rachlow and Waits, 2005). Habitat fragmentation can lead to loss of allelic diversity through increased inbreeding and reduced effective population size as a result of the genetic isolation of populations. Specifically, inbreeding may result from increased self-pollination, or where remaining trees are related through recent common ancestry (biparental inbreeding; Young, Boyle and Brown, 1996). Genetic isolation and inbreeding can lead to reduced fitness or inbreeding depression through: (1) lack of effective fertilization; (2) expression of deleterious alleles (Sork *et al*., 2002); and (3) general reduction in heterozygosity (Ellstrand, 1992). Inbreeding may have especially dire consequences for species that were mainly outcrossing, such as many tree species (Ellstrand, 1992; Husband and Schemske, 1995; but see Williams and Savolainen, 1996; Young, Boyle and Brown, 1996).

Importantly, ecological factors sometimes pose a more imminent conservation threat than genetic degradation (Caughley, 1994). For example, global declines in pollinators, associated with land-use change (Ricketts, 2001; Steffan-Dewenter, Munzenberg and Tscharntke, 2001; Baum *et al*., 2004; Kremen *et al*., 2007) and fragmentation (Frankham, 1995; Eckert *et al*., 2010), may disrupt mutualistic relationships and constitute a problem not only for species survival, but also for continued ecosystem function and crop production (Biesmeijer *et al*., 2006). If ecological or demographic risks are the most pressing, an undue focus on the genetic consequences of fragmentation can represent a missed opportunity to address key ecological risks and environmental factors of immediate concern (Asquith, 2001).

Generalizations about the potential genetic effects of fragmentation must be evaluated in the light of evolutionary history, life history and mating systems to provide a more complete, albeit complex, understanding (Kramer *et al*., 2008). In this context impacts on pollinators are as important to understanding genetic impacts of fragmentation on trees. Studies of fauna and flora suggest that some species appear more vulnerable to fragmentation than others. For example, species with large range area requirements, primary habitat specialists (Tilman *et al*., 1994; Laurance *et al*., 2001) and those with low population growth rates or poor dispersal ability may be especially vulnerable. Similarly, species with low population density, such as tropical trees, may be more vulnerable because populations are already small in number and spatially diffuse, although these species may also have adaptations that allow persistence at low density, such as pollination mechanisms adapted to obligate long-distance pollination (Kramer *et al*., 2008). Some species of both fauna and flora are also particularly vulnerable to edge effects, where land at the edge of the habitat patch is altered and the environment becomes more extreme and less amenable (Woodroffe and Ginsberg, 1998).

Pollen flow is the primary mode of gene flow in plants (Ellstrand, 1992; Young, Boshier and Boyle, 2000; Slavov, DiFazio and Strauss, 2002; Bittencourt and Sebbenn, 2008) and knowing how this changes as a result of fragmentation is vital to understanding fragmentation impacts on trees. Generally in plant species, levels of pollen flow between fragments appear to be affected by interspecific differences in longevity, generation time and pre-fragmentation abundance, the range of sexual and asexual reproductive systems (Young, Boyle and Brown, 1996; Cascante *et al*., 2002; Kolb and Diekmann, 2005), habitat specificity (Davies, Margules and Lawrence, 2000), plant height (Kolb and Diekmann, 2005), pollination and seed dispersal syndromes. Studies also show that the impacts of fragmentation on pollen flow are more varied, complex and subtle than original theoretical predictions. Given this complexity, it is unsurprising that many studies have found small or no clear genetic effects. Kramer *et al.* (2008) suggest that four key assumptions in fragmentation studies must be re-evaluated: (1) fragment edges delimit populations; (2) genetic declines manifest quickly enough to be detected; (3) species respond similarly to fragmentation; and (4) genetic declines supersede ecological consequences. For tree species, for example, the assumption that pollen dispersal stops at fragment edges is contradicted by evidence that in many cases pollination between fragments is not at all rare (e.g. Young and Merriam, 1994; Nason and Hamrick, 1997; Dow and Ashley, 1998; Streiff *et al*., 1999; Apsit, Hamrick and Nason, 2001; White, Boshier and Powell, 2002; Latouche-Halle *et al*., 2004; Nakanishi *et al*., 2004; Lander, Boshier and Harris, 2010). Thus, quite ordinary pollen dispersal may be sufficient to link trees in scattered forest fragments into a functioning metapopulation. In this case the potential negative genetic effects of small population size would not be realized. This positive view must be balanced by evidence of altered patterns of pollen flow, whereby connectivity is maintained but biparental inbreeding increases or reduced pollen pool diversity is sampled in mating. Genetic signals may also require several generations to appear, which could amount to hundreds of years in the case of long-lived tree species (Kramer *et al*., 2008). Moreover, even if forest fragments are not reproductively isolated or suffering immediate losses of genetic diversity, there may be quantitative pollen limitation (O'Connell, Mosseler and Rajora, 2006) or seed dispersal limitation, which could limit recruitment (Kramer *et al*., 2008).

4.2. Management of fragmented landscapes

Protected areas

Worldwide, countries have designated protected areas to conserve predominantly terrestrial native ecosystems and biodiversity features (Mace *et al*., 2005). Biomes differ widely in the percentage of total area under protection. Of the lands classified in the four highest IUCN protection categories, flooded grasslands, tundra, temperate coniferous forests, mangroves and boreal forests have the highest percentage area under protection. This may be because these biomes are among the least useful for competing land-uses such as agriculture. Temperate grasslands, Mediterranean forests and tropical coniferous forests are the least protected biomes. Many protected areas exist within landscapes that are fragmented to a greater or lesser degree, and may not be large enough to be viable in the long-term. As such, despite their protected status they are subject to the same biological issues that face any remnant fragment of a native ecosystem.

A continuing debate relating to reserve design is whether it is biologically more effective to set aside a single large reserve area or several small ones (Diamond, 1975). Generally, reserve design has been based on theoretical estimates of extinction risk and colonization rates and, most practically, land availability. Population dynamics models suggest that the reserve design that minimizes extinction risk is species- and case-specific, depending on dispersal ability, environmental factors, and extinction and colonization patterns (McCarthy *et al*., 2011). However, most models do not take into account the influence of uncertainty in extinction risk on optimal reserve design. Mathematically, rather than minimizing the expected extinction risk, a better objective may be to maximize the chance that extinction risk is acceptably small (McCarthy *et al*., 2011). In practice, the creation of reserves remains limited by land availability, resources to manage reserves, local support and participation, continuity of political will and capacity to protect reserve areas, and competition for other land-uses, such as food production. This has usually resulted in a bias of protected areas to upland areas and an absence on lowland fertile soils. Coupled with deforestation and fragmentation, often superimposed on habitat heterogeneity, the result is a disproportionate loss of ecosystems, species, populations and genotypes adapted to lowlands and fertile soils.

The shortcomings of conservation methods that rely on exclusion of people from reserves are increasingly recognized. Problems stem from land-tenure conflicts, displacement of local people and/or their activities and development needs, the costs of reserve management and protection, and opportunity costs for countries where reserves are located (Wells and Brandon, 1992; Brockington and Schmidt-Soltau, 2004). To mitigate these problems some initiatives have experimented with allowing human activities inside reserves or inside buffer areas around reserves. Other initiatives have taken the view that if local people benefit from the reserve they will be motivated to protect it and so have actively encouraged local people to use reserves; this has been called conservation through use (CTU). However, after 15 years there is limited evidence that CTU initiatives achieve species and ecosystem conservation at the same time as improving local livelihoods (Belcher and Schreckenberg, 2007; Barrance, Schreckenberg and Gordon, 2009). A multidisciplinary approach is needed to investigate the potential for integrating conservation and development and, more specifically, which species are or could be sustainably conserved in such systems, from both biological and human management perspectives. Efforts to maintain genetic diversity and adaptive capacity within species are irrelevant if current management drastically reduces the possibility of population persistence.

Remnant trees

As discussed above, habitat fragmentation and physical isolation do not always impede pollen flow and may increase it (but see Cascante *et al*., 2002; Hamrick, 2004; Sork and Smouse, 2006; Byrne *et al*., 2007). Despite large-scale studies, we still do not know at what distance forest fragments become genetically isolated, although new research is showing that the question is perhaps increasingly irrelevant. Rather than complete isolation, evidence currently points to alterations of mating patterns with increased distance. In some cases, single or "isolated" trees receive pollen from a wide spatial and genetic array of pollen donors, and more individual pollen donors than trees in groups (Hamrick, 2004). Single trees may also be less likely to receive pollen from their nearest neighbours than trees in groups (Chase *et al*., 1996; White, Boshier and Powell, 2002; Dick, Etchelecu and Austerlitz, 2003), although in other species the reverse appears to be true (Ward *et al*., 2005). The extent to which such single trees exhibit selfing appears to depend on the presence and strength of any self-incompatibility system within the species. Other paternity studies in fragmented landscapes have shown that, while the majority of pollen dispersal events are of the order of tens or hundreds of metres in both wind-pollinated (e.g. Dow and Ashley, 1996; Sork *et al*., 2002) and insect-pollinated species (e.g. Kwak, Velterop and van Andel, 1998; Konuma *et al*., 2000; Lander, Boshier and Harris, 2010), pollen can travel tens or hundreds of kilometres (e.g. 6–14 km in *Ficus* spp., Nason and Hamrick, 1997; Nason, Herre and Hamrick, 1998; 3.2 km in *Dinizia excelsa*, Dick, Etchelecu and Austerlitz, 2003; see also Kwak, Velterop and van Andel, 1998; Chuine, Belmonte and Mignot, 2000; Hamrick and Nason, 2000; Burczyk, Lewandowski and Chalupka, 2004). In addition, pollen dispersal distances have been shown to exceed pollinator flight distances as a result of pollen carryover (Ellstrand, 1992; Ghazoul, Liston and Boyle, 1998). Such data suggest that remnant trees and small patches of trees can be effective and important in maintaining genetic connectivity across fragmented landscapes and in conserving genetic diversity (Lander, Boshier and Harris, 2010).

Corridors

As a result of the dominance of island biogeography-based ideas that (1) habitat and non-habitat are clearly distinguishable and (2) non-habitat is wholly hostile for organism travel (MacArthur and Wilson, 1967; Ricketts, 2001; Vandermeer and Carvajal, 2001; Jules and Shahani, 2003), management of fragmented landscapes has frequently focused on the impacts of spatial isolation of individuals or species (Sork and Waits, 2010). The land between habitat patches has been considered ecologically uniform and generally hostile (Ricketts, 2001; Vandermeer and Carvajal, 2001), with the probability of organism survival and dispersal treated as a function of habitat fragment size and linear distance between fragments (isolation by distance; Jules and Shahani, 2003). Given this focus, research is frequently framed in terms of how probable it is that an organism will be able to pass through a certain area to move between habitat patches.

Programmes to mitigate the potential negative effects of fragmentation have tended to focus on increasing landscape "connectivity," defined as the degree to which the landscape facilitates or impedes movement of organisms between habitat patches (Adriaensen *et al*., 2003). Increasing connectivity between habitat patches is expected to increase effective population size, reduce inbreeding and facilitate migration, dispersal and colonization (Li *et al*., 2010; Hagerty *et al*., 2011). In a landscape classified in a binary way into habitat and non-habitat, the logical approach to increasing connectivity between habitat patches is to build bridges, called corridors or stepping stones (Villalba *et al*., 1998; Adriaensen *et al*., 2003). Corridors are narrow strips of habitat built or conserved to connect habitat patches, while stepping stones are small patches of habitat scattered through the non-habitat area between larger patches of habitat (Levin, 1995; Nason and Hamrick, 1997; Lowe *et al*., 2005). The hypothesis is that the more similar the corridor area is to native habitat, the more likely it is that an organism will move through it. Although much theoretical and empirical attention has been given to biological corridors and stepping stones, whether they are effective or not is unresolved (e.g. Simberloff *et al*., 1992; Pullinger and Johnson, 2010; Richard and Armstrong, 2010) and may often be related to a lack of clarity over identification of the target species and their specific connectivity needs.

Conservation outside protected areas: an integrated landscape approach

There is a growing perception that non-habitat areas outside reserves and outside remnant patches of native habitat may provide non-ideal rather than fatal environments (Gustafson and Gardner, 1996; Moilanen and Hanski, 1998; Arnaud, 2001; Vandermeer and Carvajal, 2001; Bender, Tischendorf and Fahrig, 2003; Coulon *et al*., 2004; Lander, Boshier and Harris, 2010). Numerous empirical studies have shown that the type of non-habitat between habitat patches affects patterns of insect and other animal dispersal and hence seed and pollen dispersal (Ghazoul, Liston and Boyle, 1998; Franklin *et al*., 2000; Davies, Melbourne and Margules, 2001; Ricketts, 2001; but see Bruna and Kress, 2002; Baum *et al*., 2004; Darvill *et al*., 2006). Thus the focus of conservation management can change from (1) how much land can be set aside, (2) how to minimize the linear distance between habitat patches or (3) how to create habitat bridges, and turn instead to measures of separation between habitat patches that incorporate variation in how easily the target organism passes through the different land-use types in the matrix (i.e. permeability; Spear *et al*., 2010; Doerr, Barrett and Doerr, 2011; Hagerty *et al*., 2011; Lander *et al*., 2011).

Although the permeability concept recognizes the potential for variation in non-habitat resistance to pollinator movement, it is still based on a binary landscape model where the question is about the study species' presence in, absence from or travel between native habitat patches. Some recent models of organism movement in fragmented landscapes are not concerned with the designation of different parts of a landscape as habitat or non-habitat, but rather focus on the quantity and accessibility of resources and threats in that landscape. Thus, land outside traditionally defined native habitat ceases to be an area to pass through and is investigated in its own right for its capacity to provide habitat services as well as its ability to support or inhibit movement (e.g. Lander *et al*., 2011). The entire landscape in this case may be considered a patchwork of partial habitats of varying quality (Kremen *et al*., 2007).

A growing body of research suggests that this "partial habitat" or "resource model" view may be both accurate and useful. For example, urban gardens, rough grassland and clover leys, none of which are native habitat, provide vital habitat services for bumblebees in agro-ecosystems (Goulson *et al*., 2010). Similarly, models that incorporate resource availability in various land-use types, such as floral resources for bees in both agricultural fields and native vegetation, have high explanatory value in predicting bee abundance and species richness (Winfree, 2010). Thus, landscape models that recognize the potential habitat services that different, apparently non-habitat, land-uses may provide could be a useful basis for interpreting empirical data and developing landscape management strategies.

4.3. The use of native species in ensuring functionality in fragmented landscapes

Against this background, land managers are asked to select, design, manage and link landscapes that will be effective in conserving biodiversity. Reforestation of degraded tropical forest lands has frequently involved the establishment of single-species plantations of fast-growing exotic species (e.g. *Pinus, Eucalyptus* or *Acacia*) which generate mainly financial benefits, whereas ecological restoration that maximizes biodiversity may produce few, short-term economic benefits (Lamb, Erskine and Parrotta, 2005). If, as is often the case, restoration must be balanced with financial returns, plantations of native species, in monoculture or as species mixtures, may provide more biological value than plantations of exotic species (Chazdon, 2008) while still providing better financial returns than "pure" ecological restoration projects. The economic and biological value of plantations of native species may be increased by underplanting the trees with shade-tolerant agricultural cash crops or species that produce non-timber forest products. Compared with monocultures, mixed plantations can deliver higher production, protection against disease and pest damage (ecological resilience) and greater security in uncertain future markets (financial resilience).

In highly modified landscapes where specific corridor development is called for, studies support a broad vision of corridor design where a range of land-uses may be combined to provide a permeable and ecologically functional landscape rather than the traditional approach of building continuous habitat corridors connecting intact forest. Corridor design, management and monitoring should involve assessment of different land-use types in terms of their ability, individually and in combination, to support movement of target species and provide the resources target species need. Assessments of corridor function will be linked to the characteristics of the target species, sustainable land-use aims of the area and variability between species and ecosystems in their resilience to disturbance. The balance of land-use types may need modification to maintain or improve connectivity.

This approach to landscape management is based on identifying land-uses that provide both habitat services and social and economic returns; the balance between these needs is clearly context dependent. For example, in an area of high forest cover, land-uses may be assessed principally for gene flow, whereas in much more highly deforested areas a fuller complement of benefits may be sought from particular systems, with their specific location in the corridor zone also being important. Thus, in the highly deforested dry forest zone of western Honduras the traditional Quezungual fallow system (Kass *et al*., 1993), in which farmers manage naturally regenerated shrubs, fruit trees and timber trees among their crops, is likely to provide a variety of genetic conservation benefits for a range of native tree species without the need for establishing specific biological corridors. Other complex systems, such as traditional shaded coffee or jungle rubber, may also rate highly for genetic conservation benefits. In contrast, simpler agroforestry systems such as pasture trees and living fences offer fewer genetic conservation benefits and are unlikely to prove effective mediators of pollen flow for species without a self-incompatibility mechanism. The emphasis generally should be on maintenance and improvement of economically and socially viable landscapes that promote connectivity (for genes, species and ecological processes) and conservation of biodiversity more generally.

Importantly, assessments of the genetic conservation benefits of agro-ecosystems are more likely to be species specific than management-system specific, and need to take into account the farming system, density of trees and their origin (natural regeneration or planted). For example, maintaining native timber trees over large areas of coffee is likely to have beneficial genetic effects for gene flow, population numbers and conservation of particular populations. However, if the same system were used in only a small area it could lead to a reduced genetic base in seed production through related (biparental) mating. Thus, the area or management unit should be measured in numbers of participating households or numbers of land units in which land-uses beneficial to target species conservation are practised (Boshier, Gordon and Barrance, 2004). Given the speed with which land-use may change in response to market prices, this measure in itself may require monitoring.

Identifying the factors that leave some species genetically susceptible to human disturbance requires extensive reproductive and regeneration ecology and genetic data. The lack of information, resource limitations and the need for more immediate action in many situations necessitates pragmatic best-guess approaches to identify which land-uses may favour gene flow for which species and which will not. The ability to extrapolate from results from model species to make more general recommendations for species management groups (combining ecological guild, spatial distribution and reproductive biology) depends on the existence of basic biological information (e.g. incompatibility and pollination mechanisms, dispersal and seedling regeneration) that enables species to be classified (Jennings *et al*., 2001).

Consideration of available information suggests that the following species types are unlikely to show genetic conservation benefits from tree-based agro-ecosystems: outcrossing species that are self-compatible; slow-growing species that reproduce only when they are large (or, in the extreme, monocarpic species, i.e. those that flower only once in their life); species with poor regeneration under human disturbance; species with highly specific pollinators or seed dispersers susceptible to disturbance; rare species with low population densities; and species with highly clumped distributions. Inevitably, such generalizations will be qualified by the range of factors that have been shown to influence patterns of genetic variation in trees.

4.4. Conclusions: policy and practice

- We need clear objectives in conservation planning that clearly identify target species, ecosystems and biogeographical regions.
- We need to continue to improve our understanding of the ecology of target species and ecosystems so that it is possible to make decisions about management that will: (i) create landscapes where habitat and ecosystem services are provided alongside economic outputs; (ii) increase landscape-scale genetic connectivity between remnant populations; and (iii) maintain ecosystem, species and genetic diversity.
- Evidence suggests that for many tree species, populations and individuals, gene flow may be high across some fragmented landscapes with little apparent forest cover. The view of forest fragmentation as producing genetic isolation may be more a human perception than a true reflection of actual gene flow. It is therefore important to recognize the complementary role that maintenance of trees on farms is already playing to *in situ* conservation. Trees in a whole range of agroforestry and other land-use systems may play an important but varied role in the long-term genetic viability of many native tree species, facilitating gene flow between existing reserves, conserving particular genotypes not found in reserves, maintaining minimum viable populations and acting as intermediaries and alternative host habitat for pollinators and seed dispersers (Harvey and Haber, 1999). Underestimating the capacity of many species to persist in large numbers in these agro-ecosystems under current practices could lead to the misdirection of limited conservation resources toward species not under threat (Boshier, Gordon and Barrance, 2004). Agroforestry tree populations may represent a considerable conservation resource, which, if taken into consideration, may show that species that are currently assumed to be threatened by habitat loss are thriving (Vandermeer and Perfecto, 1997).

However, although they undoubtedly contribute to reproduction in remnant forests, the benefits and effects are more complex than predicted and vary from species to species. Uneven representation and overrepresentation in pollen pools and mating may lead to non-random mating, with reductions in genetic diversity in subsequent generations. We should not overestimate the extent to which agro-ecosystems will benefit the genetic conservation of forest tree species. In addition to some of the complications raised here, it is evident that many of the tree species found in agro-ecosystems are already present in adequate numbers in existing reserves. Similarly, some of the species threatened by low population numbers are not of the type that will easily persist in such systems. The greatest potential role of agroforestry and other agro-ecosystems will be in highly deforested areas where reserves are very small or nonexistent and where the trees maintained in these systems represent an important part of a particular population's or species' gene pool. In such circumstances, the fact that many tree species that live in such disturbed vegetation can be conserved through existing practices can free resources for the conservation of more critically threatened species needing more conventional, resource intensive approaches.

- We need to ascertain which land-uses are favourable to connectivity and conservation and which are antagonistic. In the fragmented forests of central Chile, Lander *et al*. (2011) found that support for subsistence farms and modification of management to reduce the size of pine plantation clearfells would be likely to have significant positive impacts on the viability of pollinator populations and the probability of pollinators moving across the landscape between native forest fragments. Goulson *et al.* (2010) and Winfree (2010) found that urban gardens, rough grassland and floral resources in agricultural fields provided vital habitat services for pollinators in landscapes dominated by agriculture. Agricultural land itself can provide habitat services, depending on the diversity of crops, size of individual fields, use of agrochemicals, application of integrated crop- and pest-management systems and management of waterways and soils (Sustainable Agriculture Network, 2010). Although the process of identifying favourable land-uses has begun, this is a rich area of study.
- We need a broader view of conservation that recognizes that reserves can be only part of the solution to our conservation concerns and embrace the possibility that anthropogenic land-uses may provide valuable and necessary ecosystem and habitat services. In the binary view, where only native habitats, or those land-uses most similar to native habitats, are recognized as providing ecosystem services, land managers will tend to focus on distances between habitat patches, degree of habitat patch aggregation and corridor-type connectivity (Doerr, Barrett and Doerr, 2011). This lack of differentiation between non-habitat land-use types limits management options and contributes to polarization of the conservation debate, leaving decision makers with the unenviable task of choosing between economic activity or setting aside land for conservation. If we move beyond the expectation that organisms will move in a directed manner between areas that have been designated as habitat and focus instead on the ecological requirements of target species or the ecological attributes of the land-uses in the wider landscape, we may understand how best to manage a mosaic of habitats of varying quality. This type of landscape management strategy could be both more effective biologically and less expensive than traditional conservation based on land set aside for conservation.
- The complementary benefits of different land-use practices for genetic conservation must be further evaluated, recognized and promoted. There is a need to raise awareness among development professionals of the value of natural regeneration as both a conservation and socioeconomic resource. The emphasis on a limited range of species, often exotics,

by development agencies may reduce the potential genetic benefits of such systems, besides creating potential problems of invasiveness. However, there is also a need for conservation planners, more accustomed to *in situ* methods, to consider the possibility that tree populations found outside protected areas have a role in biodiversity conservation (Boshier, Gordon and Barrance, 2004). This in turn requires the direct involvement of development organizations in biodiversity conservation and an effective interaction between them and traditional conservation organizations to ensure both conservation and development benefits.

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Chapter 5

Gene flow in the restoration of forest ecosystems

Leonardo Gallo and Paula Marchelli

Unit of Ecological Genetics and Forest Tree Breeding, INTA Bariloche, Argentina

In forest trees, as in all organisms, new genetic variants are generated by mutation. The adaptive value of the new variants is initially tested by strong selection pressure during the production of sporophytes and gametophytes in the local environmental conditions in which the mutations appeared. However, the potential benefits of genetic mutations can be tested under different environmental conditions through gene flow.

Of the five main evolutionary forces (mutation, recombination, selection, genetic drift and gene flow), gene flow is the only one that can generate new genetic variation through the direct or indirect combination of genetic variants and occurs at a landscape scale. Gene flow in sessile, long-lived organisms like trees depends strongly on the movement of gametes in the form of pollen grains (pollen flow) and zygotes, usually as seeds (seed dispersal). Several tree species can also propagate vegetatively through broken twigs, root suckers or layers, distributing genetic information identical to that of the original tree. In most species pollen, seeds and vegetative propagants are moved by vectors, such as wind, animals, water or human beings.

Gene flow is defined as "the proportion of newly immigrant genes moving into a population" (Endler, 1977). Movement of genes within populations is termed gene movement (Devlin, Roeder and Ellstrand, 1988). However, anthropogenic impacts have modified the movement of genetic variants not only among populations but also, frequently, within them. Moreover, the physical dimension of a "biological population" (individuals that exchange genetic information and share a common evolutionary path) is difficult to define in nature. As a result, for practical purposes, such as for restoration, gene flow is considered to include not only exchange of genes among populations but also the local reproductive system dynamics within them.

We differentiate between "gene flow" and "effective gene flow" (see Lowe, Harris and Ashton, 2004). Effective gene flow at the pollen level is influenced by movement of the pollen grain from the male flower to the female flower, germination in the pistil or equivalent structure and fertilization of the ovule to form a new zygote (seed). Pollen can move between flowers of the same tree, between flowers of different trees within the same population and/or between flowers of different trees from different populations. Effective gene flow at the seed level is influenced by the movement of the seed, its germination and establishment as a sapling. Seed can be dispersed to different places within the same population, to different populations and/or to newly available habitats (i.e. colonization).

Pollen flow is much more extensive than seed dispersal (see, for example, Petit *et al.*, 2005). However, it has been shown that seed dispersal can be more effective than pollen dispersal at maintaining genetic connectivity in exceptional circumstances, as in the case of *Fraxinus excelsior* fragments in an ancient deforested landscape (Bacles, Lowe and Ennos, 2006).

5.1. Genetic effects at different scales

Gene flow connects populations and influences colonization, and therefore constitutes one of the main processes to be considered when managing and/or restoring forest ecosystems. Human activity continuously modifies the environment at various overlapping spatial and temporal scales. Matching the two dynamic processes of effective gene flow and environmental change is essential to promote viable and adaptable forest ecosystems.

Gene flow integrates many different elements and functions of the ecosystem through various selection gradients. Any restoration activity should take into consideration the complex equilibrium that rules gene flow in the desired species. Assessment of this equilibrium should be made at the appropriate spatial and temporal scales, given that effective gene flow in tree species occurs over ranges from tens of metres to several kilometres and generation intervals are long.

The general increase in genetic diversity within populations due to gene flow can be considered as an advantage for the adaptation and adaptability of the forest ecosystems. This is especially true when the immigrant genes and/or genotypes are better adapted to the local environmental conditions than the local genotypes. However, gene flow can introduce undesirable genes and/or genotypes if the local population is already well adapted and in evolutionary equilibrium with the environmental conditions. If the influx of maladapted genes is large relative to the size of the extant population, the local genetic adaptation can be undermined, although natural selection would be expected to weed out poorly adapted individuals over time. However, the long-distance gene flow commonly found in forest tree species may augment the ability of populations to respond to climate change through a general increase in genetic variation in the population (Kremer *et al.*, 2012).

5.2. Considerations in restoration and management

Forest restoration and management activities may have an active or passive impact on the gene flow occurring in the system. Active gene flow management (AGFM) relates to the movement of genetic material from one location to another by human beings. Passive gene flow management (PGFM) relates to the modification of the landscape and environment to facilitate the natural movement and recruitment of genetic variants into the population to be restored. It could include, for example, the reintroduction of native animals functioning as pollen and/or seed vectors. Both types of interventions can and should be used at the same time in some restoration situations. In PGFM, human activities modify not only the landscape in which natural gene flow takes place and/or new genotypic variants establish but also the local environmental conditions. In some cases the improvement of the receptor environment could facilitate effective natural dispersion by increasing the likelihood of establishment of the desired genetic variants.

In some special cases, AGFM can be implemented even across continents. In the last century, southern beech species (*Nothofagus* spp.) originating in Chile and Argentina were used to restore some English landscapes because they grow better than native Fagaceae species (L. Gallo, personal observation), while maintaining landscape functionality and visual appearance (Poole, 1987). A female clone of *Salix* × *rubens,* a hybrid between *S. fragilis* and *S. alba*, was introduced into the Patagonian region of South America from Eurasia and over the last 100 years has colonized huge areas, in some cases more than 100 000 km^2, mainly through natural distribution of broken twigs by water (Budde *et al.*, 2011). It has also hybridized and introgressed with the native species, *S. humboldtiana*, competing for its natural habitat and diluting its gene pool (Bozzi *et al*., 2011). Another remarkable case of human influence on the distribution pattern of genetic diversity in forest tree species is the vegetative propagation and distribution of an elm clone of *Ulmus minor* var. *vulgaris* (*Ulmus procera*) by the Romans, who used it to support grape vines in their vineyards in France, Spain and England (Gil *et al.,* 2004). *Araucaria araucana* was probably introduced in some areas of northern Patagonia by the Mapuche people, who used its seeds as food (Gallo, Letourneau and Vinceti, 2004; Marchelli *et al.*, 2010).

At the landscape scale, the effect of human activities on gene flow is a consequence of forest ecosystem modifications mainly through fragmentation and introduction of artificial barriers. Both of these mainly affect vectors.

The genetic consequences of habitat fragmentation depend on whether it affects gene flow; if it restricts gene flow, fragmentation is highly deleterious in the long term (Frankham *et al.,* 2002). When fragmentation occurs, movement of pollen and seed is determined by the distances between the fragments (particularly in wind-pollinated species) or the environmental conditions of the landscape between the fragments (particularly in insect-pollinated species). The effects of forest fragmentation on the behaviour of pollinators and animal vectors that disperse seed differ depending on species and cannot be generalized. In some cases fragmentation can reduce gene flow distances (Powell and Powell, 1987) and in others has been shown to increase gene flow (White, Boshier and Powell, 2002).

In some regions of the world gene flow is affected by unidirectional movement of the vector during pollination or seed dispersal. In such cases the effect of fragmentation depends essentially on the location of the fragments. This is the case for fragmented populations of Patagonian Cypress, *Austrocedrus chilensis*, growing in a xeric region where less than 2 percent of the wind blows a north–south (or south–north) during pollination and more than 75 percent blows west–east (Figure 5.1). This dioecious species has been shown to have pollination distances of over

Figure 5.1.
Fragment of *Austrocedrus chilensis* (Patagonian cypress) with about 100 hundred individuals, separated from a neighbouring fragment by just 1200 m and occurring on an arid grassland steppe matrix with 350 mm of mean annual precipitation. The orientation of the fragments is north–south and therefore gene flow is severely restricted since wind persistently blows in west–east.

1000 m in the fragmented margins of it s natural distribution (Colabella, 2011). However, fragments lying less than 1200 m apart on a north–south axis were found to be genetically isolated from each other using isozymes (Gallo and Pastorino, 2010) and microsatellite markers (Arana *et al.,* 2010). In addition to the reproductive isolation, the small effective population size of these fragments resulted in genetic drift that is expressed in the fixation and loss not only of some neutral alleles but also of some adaptive and continuously varying traits. In this case the vector, namely the wind, acts as a dynamic barrier because of its persistent directionality ("reproductive isolation by wind," Gallo and Pastorino, 2010).

Poverty and lack of ecosystem protection and management controls affect the pollen and seed dispersal of many forest tree species. For example, hunting of animals that are seed or pollen vectors reduces their population size and therefore reduces gene flow. In contrast, introduction of novel vectors can offset other impediments to gene flow. For example, introduction of African bees, which can fly long distances in fragmented landscapes, resulted in greater pollen flow between *Dinizia excelsa* (Fabaceae) trees in fragmented Amazonian rainforest than that recorded in pristine forest without the African bees (Dick, Etchelecu and Austerlitz, 2003). In some wind-pollinated species, fragmentation increases the speed and free movement of wind and consequently the dispersal distances of pollen and seeds (Young *et al.*, 1993; Bacles, Lowe and Ennos, 2006).

Gene flow can also be restricted or completely interrupted by artificial physical barriers (plantations of introduced species, buildings, windbreaks etc.). However, in Australia significant gene flow has been reported between remnant natural populations of *Eucalyptus loxophleba* and introduced plantations of the same species (Sampson and Byrne, 2008). In the Canary Islands, the natural regeneration in artificial plantations of *Pinus canariensis* were found to have greater genetic diversity than the planted adult trees as a result of pollen flow coming from surrounding natural stands of the same species (Navascués and Emmerson, 2007).

As stated before, effective gene flow requires the recruitment of the transported genetic variants in the new environment, and habitat disturbances can impede or facilitate this process. For example, despite evidence of extensive pollen flow between western continuous forests of *Araucaria araucana* and fragmented eastern populations (Gallo *et al.*, 2004), no regeneration was found in several of the fragments because of the severity of anthropogenic impacts, particularly the impacts of grazing livestock, animals introduced for hunting and collection of seed by humans. Thus, *effective gene flow* was zero and the sustainability of the whole system is threatened although gene flow between continuous and fragmented populations exists (Gallo *et al.*, 2004).

A very well known case in which human activity affects gene flow among plant species is the creation of environments that encourage the development of interspecific hybrids (Arnold, 1997). In such human-induced "hybrid habitats" the parental species can barely survive but the interspecific hybrids thrive (e.g. Campbell and Wasser, 2007). In many cases, gene flow between the parental species could not have been realized without the environmental disturbance, as has been shown in *Prosopis chilensis* and *P. flexuosa* (Mottuora *et al.*, 2005).

Gene flow is strongly related to the mating system and therefore to the fitness of individuals and populations. Many forest tree species have a very strong spatial genetic structure, even in large, continuous forests. Related individuals tend to grow in groups because of limited dispersal of seeds around the mother tree and/or spatial directionality of the vectors. This structure depends mainly on two opposing forces: selection pressure and gene flow. The natural vectors of pollen and seeds determine how large the realized mating system can be. Human activities modify both: the spatial structure of the remaining adult genotypes defining the distances between them and the environment in which seed has to germinate and seedlings have to establish. In spe-

Box 5.1.
Fundamental considerations for gene-flow management in restoration activities

Effective gene flow

Gene flow should be evaluated on the recruited regeneration after passive or active restoration. The following considerations therefore depend on there being effective gene flow, i.e. establishment and evolutionary adaptation of the genetic variants.

Genetic diversity

Only genetic diversity can assure current adaptedness and future population adaptation. Therefore, its magnitude and distribution have to be known in both the population to be restored and the population or trees to be used as donors of genes and/or genotypes (propagation material). If some fragments are subjected to genetic diversity restrictions as a result of demo-genetic processes (bottle necks, genetic drift, biparental inbreeding etc.), material from other populations with more genetic diversity (preferentially from the same seed transfer zone) should be used to assure evolutionary stability.

Genetic connectivity

Restoration activities must ensure the movement of genetic information between trees or fragments within a biological population (individuals that share an evolutionary path). Dispersion curves and dispersal distances for pollen, seed or vegetative parts must be estimated. New molecular methods are increasingly more precise, cheaper and easy to apply than traditional methods and facilitate this work. The effect of potential physical barriers established by humans (e.g. forestry) should be considered when managing passive gene flow for genetic connectivity.

Climate-change scenarios for the restoration area

In the current global scenario of climate change, even well-adapted populations should be restored using material from populations that are better adapted to expected future environmental conditions at the location being restored. In some cases active or passive gene flow might be promoted, in general, from drier environments towards more humid sites.

Isolated trees

In highly fragmented landscapes efforts should be made to maintain existing isolated trees and to integrate them in the local socioeconomic system, since they act as genetic bridges (receiving pollen and dispersing seeds) and as ecological corridors for pollinators between fragments and/or continuous forests. Knowing the pollen movement distance of the species involved allows development of a network of isolated trees that can help maintain the genetic connectivity in such agricultural landscapes.

Unidirectionality of vector movements

The relative location of the fragments restored must be considered, especially with wind-pollinated species. In insect-pollinated species "attractor trees" (e.g. abundantly flowering trees for generalist insects) should be established at strategic locations within the agricultural landscape to guide the pollinators' movements.

Impact on animal vectors

Many animals act as vectors of pollen or seeds, especially in tropical and subtropical forests. When restoration activities are implemented, illegal hunting activities should be forbidden and traditional hunting activities of local communities should be organized and controlled to ensure that the vector population is maintained at a viable level. When restoration activities are undertaken in strongly fragmented agricultural landscapes, special consideration should be given to managing the use of pesticides and herbicides in the surrounding cultivated areas. The type and amount of chemicals to be used, the timing of their application and location of buffer zones should be agreed with local communities and/or agricultural enterprises to minimize negative impacts on pollinator-insect populations.

Stand genetic structure

Sustainable stand density should be taken into account when restoring degraded forest populations. Effective population number and probable biparental inbreeding has to be monitored when passive restoration strategies are implemented.

Box 5.1. *(continued)*
Fundamental considerations for gene-flow management in restoration activities

Hybridization
Active restoration activities can unintentionally introduce material from high genetically differentiated populations into the population to be restored, altering the natural gene-flow equilibrium. The potential effects of hybridization between remnant and introduced material have to be considered since outbreeding depression may have negative effects on adaptation.

cies with very short pollination distance, management practices can severely alter gene flow and through it the mating system. For example, pollen of the southern beech (*Nothofagus nervosa*) effectively disperses less than 35 m but an average of nine pollen parents contribute to each mother tree, indicating that density of the stand is crucial for the movement of the pollen (Marchelli, Smouse and Gallo, 2012). When forest management activities reduce stand density, a reduction of the genetic diversity in subsequent generations of the managed forest is expected, mainly due to the increase of biparental inbreeding.

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Chapter 6

The role of hybridization in the restoration of forest ecosystems

Leonardo Gallo

Ecological Genetics and Forest Tree Breeding, INTA Bariloche, Argentina

Restoration of genetic diversity implies the management of different levels of genetic exchange, including hybridization.[10] The most common understanding of hybridization is mating between related species, but consequences of intraspecific hybridization must also be considered. When successful mating occurs naturally between individuals from two populations, or groups of populations, that are distinguishable on the basis of one or more heritable characters, natural hybridization takes place (Harrison, 1990). One of the main potential evolutionary outcomes of intra- or interspecific hybridization is introgression, which means the movement of the genes from one population or species into the other as a result of successive backcrosses (Anderson, 1949). But there are other potential theoretical outputs, such as increased genetic diversity through the generation of new gene combinations and genotypes, heterosis, hybrid speciation, reinforcement of the reproductive barriers that favour parental speciation, and stabilization of hybrid zones (Carney, Wold and Rieseberg, 2000). Recently, hybridization has been highlighted as a way to regain traits that had been lost and perhaps to replace damaged alleles with functional copies from related species (Rieseberg, 2009). An often mentioned negative consequence of hybridization is the genetic dilution of a rare population through introgression and exogamic or outbreeding depression caused by dilution of the local adaptation and hybrid breakdown (e.g. disruption of well co-adapted gene complexes) (Hufford and Mazer, 2003). Additionally, in many hybridizing systems a strong environmental influence has been observed in the hybrids' generation and fitness. Ecosystem degradation alters environmental conditions and consequently affects some biologically important traits such as gene flow, gamete production and flowering phenology that promote hybridization (e.g. Lamont *et al.*, 2003; Mottuora *et al.*, 2005). In those altered "hybrid environments" hybrids have an adaptive advantage over their parents (Arnold, 2006).

6.1. The impact of restoration

Restoration activities can impose genetic connectivity by moving seeds and establishing plants from the donor population directly into the degraded population or by favouring gene flow between them (active or passive restoration), creating conditions for hybridization between populations or species that were not previously in contact. This can have positive or negative impacts, depending on the situation.

During the genetic restoration process the occurrence of natural hybrids can be promoted or avoided, depending on their expected fitness, the degradation level of the ecosystem and the final objectives of the restoration.

[10] Hybridization in the text is taken in a wide sense and as a long term process including F1, F2, backcrosses, etc., among different populations of the same species and/or of different species.

6.2. Promoting hybridization

Under very strong selection pressures, increased genetic variation and/or the generation of new genotypes through hybridization can be adaptively advantageous. For example, a controlled introgression programme has been implemented to rescue the genetic information and the ecological and economic benefits of American chestnut (*Castanea dentata*), which was devastated by an exotic pathogen, chestnut blight (*Cryphonectria parasitica*), at the beginning of the twentieth century. A programme of controlled hybridization and introgression incorporated resistance genes from the Chinese chestnut (*Castanea mollisima*) into the genome of the American chestnut with remarkable success. Recently, candidate genes for developing resistance have been identified through the use of advanced molecular technologies (Barakat *et al.,* 2009).

Controlled hybridization programmes may become important means to confront climate change and counteract the negative effects of drought. Several genetically different donor populations having drought tolerance could introduce into the degraded ecosystems potentially genes that could confer better adaptation to future climatic conditions.

6.3. Avoiding hybridization

If the divergence between the hybridizing populations is caused by differences in local selection (local adaptation), maladapted hybrids would be expected (Hufford and Mazer, 2003) and hybridization should be avoided or mitigated.

Such maladaptation has been observed in the natural hybridization between two Patagonian southern beeches, *Nothofagus nervosa* and *N. obliqua* (Gallo, Marchelli and Breitembücher, 1997), where selective logging in the past had substantially reduced the population of *N. nervosa* and altered the natural pollen competition equilibrium, promoting the generation of maladapted hybrids (Gallo, 2004). First-generation hybrids have reduced fitness and the system has reached a particular equilibrium where backcrosses are also limited (the "evolutionary novelty" hybridization model described by Arnold, 1997). In such situations, restoration activities should include the reconstruction of the original species structure of the forest.

If hybrids are well adapted, a large proportion of the gametes produced by the few individuals of the degraded population will generate hybrids and through introgressive backcrosses their genetic information will tend to be diluted; a process known as genetic assimilation. At a regional scale, this is the problem with the European black poplar (*Populus nigra*) and the eastern cottonwood (*Populus deltoides*), introduced from the United States. Genetic rescue activities have been carried out to save the few pure individuals of black poplar since the hybrid (*P.* × *canadensis*) competes for river niches and introgresses its genetic information into the black poplar (Smulders *et al.*, 2008). Such a situation can be also expected in intraspecific hybridization when restoring degraded populations.

Landscape fragmentation can increase gene flow, depending on the species' pollination mechanisms, and with it the negative genetic exchange of diverging populations. The use of physical barriers (intermediate forestations) or even removing the contaminating trees could be possible solutions.

6.4. Seed sources and seed-zone transfer

When gene flow is restricted among isolated fragments in a heterogeneous landscape, the occurrence of strong local adaptation processes and/or genetic drift divergence has to be considered. In such cases, the use of local propagation material is recommended for active restoration programmes.

Gene flow and fluctuating selection pressure can reduce the probability of development of highly localized ecotypes (McKay *et al.*, 2005), especially where there is extensive pollen flow in long-lived species (Moreno, 2012). Moreover, in many ecosystems site disturbance introduces a new challenge for local adaptation and restoration (O´Brien and Krauss, 2010), to which global climate change impacts should be added. Seed-zone transfer guidelines should be determined.

To minimize the probability of outbreeding depression, and when lacking a scientifically-based delineation of seed zones, seed collections should be made near the restoration site, if populations of sufficient size and genetic quality are available as seed sources (Hufford and Mazer, 2003), in order to match climatic and environmental conditions (McKay *et al.,* 2005).

The combination of a network of common-garden field trials across a range of representative site conditions with molecular genetic diversity and gene flow analyses could help to better determine operational genetic management units (Pastorino and Gallo, 2009) that should be taken into account in any restoration activities.

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Chapter 7

Collection of propagation material in the absence of genetic knowledge

Gösta Eriksson

Department of Plant Biology and Forest Genetics, Swedish University for Agricultural Sciences, Sweden

Genetic variation within and among populations has not been studied in the vast majority of tree species. This makes it difficult to plan effective germplasm collection strategies for forest restoration and species conservation purposes. This paper provides guidelines for collection of propagation material for forest restoration when knowledge of genetic variation between and within populations is lacking. Rare species that occur as scattered trees or in small cohorts growing hundreds of metres apart are unsuitable sources of propagation material for stand establishment since they never form stands in nature. Therefore, this paper refers to commonly occurring species.

Forest restoration projects may also aim to conserve the genetic diversity of the species used, but for a specific treatment of sampling for gene conservation in *sensu stricto*, the reader is referred to Eriksson (2005a). Evolutionary factors are presented to help understand the existing genetic variation between and within populations.

7.1. Evolutionary factors

In nature there is constant interaction among evolutionary factors, with the result that it is hard to know or predict the genetic variation in a species (Mayr, 1988; Eriksson, 2005b). Evolutionary factors are briefly discussed here to elucidate their role in evolution. (See Box 7.1 for definition of terms.)

Natural selection requires that there is genetic variation for traits contributing to fitness (Endler, 1986). Changes in gene frequencies are dependent on existing conditions; future conditions have no influence over them. Thus, there is no goal or predetermined direction of selection. Moreover, most fitness traits are complex. For a tree they may consist of growth rhythm, growth rate, tolerance of adverse climatic factors, tolerance of pests or diseases, and uptake and utilization of nutrients. It is highly unlikely that natural selection influences these components individually; rather it is the whole individual that is the "target" in natural selection. For most traits of significance in evolution we find a bell-shaped curve for the distribution of individuals. In many cases of natural selection the individuals in the centre of the distribution are favoured (stabilizing selection) but in some cases the individuals in one of the tails of the distribution are favoured (directional selection). Selection that favours individuals in the two tails of the distribution is known as disruptive selection. This type of selection probably occurs rarely within populations.

Genetic drift causes random losses of genes, the rate of loss increasing with decreasing number of reproductive individuals in a population. At a constant population of ten fruiting trees for ten generations genetic variation (or more correctly "additive variance") falls to approximately half of the original (Eriksson, 1998). A population of 20 individuals would lose approximately

20 percent of its additive variance. The difference in loss between populations of 500 and 1000 is only 0.5 percent per generation. This means that not much is gained by having thousands of fruiting trees in gene-resource populations. A population of 50 fruiting trees has been regarded as a satisfactory size for a gene-resource population and for sourcing propagation material. With such a population size the loss of additive variance is just 1 percent per generation.

Gene flow caused by pollen and seed transfers between populations can be considerable in wind-pollinated species but less in species with limited pollen and seed dispersal (Govindaraju, 1988). Gene flow is such a strong levelling factor that only one migrant per generation prevents differentiation in neutral genes (genes not influencing fitness) in a randomly mating population.

Mutations occur at low frequencies, and therefore do not exert any strong influence on evolution in the short term. Mutations are of great significance in the long term because they create genetic variation for the other evolutionary factors to act upon.

The impact of within-population and between-population variation of natural selection, genetic drift and gene flow are summarized in Table 7.1. Stabilizing natural selection within populations leads to less overlapping of the adjacent populations. This in turn means that there will be a sharpening of the differences between populations. Since the loss of genes as a result of genetic drift is random, different genes will be lost in different populations, which also leads to increased variation between populations. The effect of gene flow on variation within and between populations is evident from the definition of gene flow. Thus, gene flow reduces differences between populations but increases the within-population variation. Understanding the principles behind the distribution of variation helps to determine how many different populations should be considered as sources of propagation material in order to adequately capture the genetic variation of a species. It can guide species restoration efforts or help to prepare for environmental changes with diverse propagation material.

TABLE 7.1.
The impact of natural selection, genetic drift and gene flow on genetic variation within and between populations

	Variation within populations	Variation between populations
Natural selection	decrease	increase
Genetic drift	decrease	increase
Gene flow	increase	decrease

7.2. Methods for sampling diversity

Before materials are selected for forest restoration, it is useful to examine the abiotic and biotic factors at the reforestation site. Abiotic factors such as climate, nutrient availability and photoperiodic conditions might be easy to determine, while biotic factors such as pests and diseases might be hard to identify in advance. At least one, but usually more, of these factors is significant at a particular reforestation site. For example, photoperiodic conditions are of significance at high latitudes where this factor varies considerably. With only basic knowledge of genetic diversity and biotic and abiotic factors we have to rely on educated guesses to develop a sampling strategy for our target species. Generally, sampling in the absence of genetic knowledge should take place at localities with closest similarity with the reforestation sites.

7.3. Genetic variation

Hamrick, Linhart and Mitton (1979) hypothesized that life history traits such as species distribution, stage in ecosystem – pioneers versus climax species – or wind pollination vs insect pollination would influence genetic variation within and between species (Figure 7.1). However, in his

Figure 7.1.
The expected effects of life history traits on genetic variation within and between populations

Source: Hamrick, Linhart and Mitton (1979)

study of adaptive traits in several tree species, Baliuckas (2002) reported only weak support for this hypothesis. Until more research results are obtained, Hamrick, Linhart and Mitton's (1979) hypothesis may still be used to guide sampling.

Sampling existing adaptations is simple if all variation is included in every population and not between them. Then, it suffices to sample enough trees to avoid inbreeding. However, absence of genetic variation among populations is not known to occur in any tree species other than the red pine (*Pinus resinosa* Ait.).

Because the evolutionary factors act in a complex way, we have to simplify the possible effect of their interactions on genetic diversity. The effect of genetic drift can be excluded if mating is random, leaving the two opposing factors natural selection and gene flow. Figure 7.2 illustrates the possible between-population differentiation for various combinations of gene flow and disruptive natural selection. A prerequisite for

Figure 7.2.
The possible genetic variation between populations in random-mating species, as affected by varying strengths of disruptive natural selection and gene flow

differentiation between populations is that the species experiences the biotic and abiotic site conditions in its area of distribution as heterogeneous. In the absence of disruptive selection there may be some differentiation of populations for random reasons (position 1 in Figure 7.2). The larger the variation of the site conditions the greater the differentiation between populations; differentiation will be greatest in the absence of gene flow (position 2). When both gene flow and natural selection are strong, populations may still differentiate (position 3).

It is obvious that species covering large areas, such as Norway spruce (*Picea abies* L. Karst.) and Scots pine (*Pinus sylvestris* L.) in Europe or Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco) and Lodgepole pine (*Pinus contorta* Douglas) in North America, face extreme variation in site conditions. It is not only the climate that varies in their distribution areas but also soil conditions. Such a large variation in site conditions causes population differentiation (Dietrichson, 1961; Eiche, 1966; Rehfeldt, 1989; Lindgren *et al.*, 1994). Since these species also are wind pollinated they are examples of species with high disruptive selection and large gene flow (position 3 in Figure 7.2). These four species experience large differences in climate over their distribution areas. For this type of species, numerous populations should be considered as sources of propagation material for reforestation. Still larger numbers of populations have to be included if the species grow on contrasting edaphic conditions within a climatic zone.

Red pine is distributed over a large area in eastern North America without much population differentiation (Mosseler, Egger and Hughes, 1992). Strangely, there also seems to be very limited variation within populations. This lack of variation has been attributed to a genetic bottleneck after the last glaciation. Red pine is an example of a species with weak disruptive selection (position 1 in Figure 7.2). Since there is one almost homogenous population, in theory seed could be collected from anywhere in the range of the species for planting anywhere. In reality, it would be prudent to exercise some caution. Gene flow does not occur in this species.

There are few examples of forest tree species that would experience natural selection in the absence of gene flow (position 2 in Figure 7.2). The Fraser fir (*Abies fraseri* (Pursh) Poir.), may have such a genetic structure. The species comprises seven populations in North Carolina, Tennessee and Virginia, United States, mainly at more than 1500 m above sea level (Pauley and Clebsch, 1990). However, a large part of the differentiation is probably due to genetic drift. For such a species, all populations should be designated as gene resource populations. Bearing in mind global warming, planting at higher elevation than the present distribution of Fraser fir is recommended if funding for such an approach could be raised. This recommendation is also valid for other species with similar characteristics to those of the Fraser fir.

Box 7.1. Definition of terms

Evolution = change of genetic constitution
Adaptation = the process of genetic change of a population, owing to natural selection, resulting in a better adaptedness
Adaptedness = the degree to which an organism is able to live and reproduce in a given environment
Adaptability = the ability to respond genetically or phenotypically to changed environmental conditions

Evolutionary factors

Natural selection = improvement of adaptedness via differential transfer of genes to the next generation
Genetic drift = random loss of genes in small populations
Gene flow = migration to a recipient population from another population with a different gene frequency
Mutation = a chemical or structural change of DNA
Random mating = each tree in a population has an equally large chance to take part in the fertilization as all other trees in this population

7.4. Avoidance of genetic drift

Once it is decided which populations should be used for collection of material, sampling should encompass enough trees to avoid narrowing the genetic variation in the stand to be established. Insufficient variation might lead to loss of the whole new stand as a result of pests, diseases or adverse abiotic factors. Because a restored stand is expected to be self-perpetuating, it is important to avoid inbreeding.

Recommendations have been formulated to guide sampling for propagation purposes (e.g. Dawson and Were, 1997). It is commonly suggested to collect germplasm from a minimum of 30 trees. To avoid offspring of related trees occurring in the sample, a minimum distance of 50–100 m between the collected trees is suggested. Broad genetic diversity of the propagation material will not only ensure a viable population, but will most likely be advantageous for adaptation to changing environmental conditions.

Selfing and other forms of inbreeding in cross-fertilizing trees cause strong inbreeding depression; stronger the closer the relatedness. Thus, selfing leads to much stronger depression than cousin matings. In one of the oldest field trials with a selfed forest tree, Norway spruce, the inbreeding depression of stem volume at 60 years of age was substantial and amounted to approximately 50 percent reduction in growth (Eriksson, Schelander and Åkebrand, 1973).

The European white elm (*Ulmus laevis* Pall.) in southern Finland is one example of a species experiencing natural selection in the absence of gene flow (position 2 in Figure 7.2). This species has high between-population variation in Finland, mainly attributed to genetic drift (Vakkari, Rusanen and Kärkkäinen, 2009). Between-population variation mediated by genetic drift does not result in high adaptedness in all small populations. To obtain good reforestation material it might be useful to put together trees in seed orchards or clone archives, as suggested for conservation of the European white elm in Finland, in which two to ten clones from each of 19 populations were planted in *ex situ* plantations for seed production (Vakkari, Rusanen and Kärkkäinen, 2009). Progenies from such plantations will be exposed to natural selection, frequently resulting in change of the genetic constitution. In this way the effects of genetic drift will be reduced and the genetic variation will be increased. The number of clones in such plantations should preferably be 50 but an absolute minimum of 20 must be met.

Considering predicted global warming, measures could be taken to mitigate the effects of a rapid environmental change (Figure 7.3). Each subpopulation should ideally consist of 50 trees. The illustrated principle may also be applied in conditions where it is desirable to support migration of a species.

Figure 7.3.
The oval area represents the current distribution of a species along a climatic gradient from a warm climate at the bottom to a cooler climate at the top. The green circles are gene resource subpopulations. Materials from the subpopulations should be transferred to a cooler climate to mitigate the impact of climatic warming. It might be wise to establish some subpopulations outside the present range of distribution (dark green circle).

7.5. Conclusion

In summary, estimates of the distribution of genetic variation within and between populations enable genetically solid conservation and also promote adaptation in species restoration efforts. Consideration of the variation between populations is especially important for species with high disruptive selection and limited gene flow (position 2 in Figure 7.2), followed by species with high disruptive selection and large gene flow (position 3 in Figure 7.2). For species with low disruptive selection and limited gene flow (position 1 in Figure 7.2), much of the genetic variation occurs within the population and less between the populations, which means that restoration material from a few populations would be sufficient.

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Chapter 8

Evaluation of different tree propagation methods in ecological restoration in the neotropics

***R.A. Zahawi*[1] *and K.D. Holl*[2]**

[1] *Las Cruces Biological Station, Organization for Tropical Studies, Costa Rica*
[2] *Environmental Studies Department, University of California, Santa Cruz, United States*

Despite ongoing pressures to clear tropical forests, there is also substantial interest in their restoration and tropical forest cover is increasing in certain regions (Asner *et al.*, 2009). The motivation for restoring tropical forests comes from an interest in enhancing or restoring the delivery of ecosystem services (e.g. sequestering carbon, minimizing erosion, improving water quality, maintaining hydrological cycling and harbouring biodiversity) and the maintenance of natural capital.

Most tropical forest restoration efforts focus on reintroducing tree species to accelerate forest recovery (Holl, 2012). There are three principal methods of tree propagation used to restore former agricultural lands in the tropics: (1) planting seedlings grown in nurseries from seed; (2) vegetative propagation of individuals, either directly onsite or in nurseries; and (3) direct seeding into a restoration site. Which of these strategies to use depends on the goals of the project, the natural rate of recovery and the ecology of the system (Holl, 2012).

Germinating and establishing seedlings from seed in nurseries is the predominant form of tree propagation and the most widely used method throughout the tropics. The two other techniques are emerging as viable potential alternatives because they are less labour-intensive and cheaper. The establishment of trees by vegetative means has typically centred on using small cuttings from young branches and shoots of trees or larger branches that are often referred to as stakes. Direct seeding refers to the mass seeding of a species or group of species into a restoration site at the onset of a project, or more-targeted seeding of typically mid- to late-successional species at later stages in the recovery process.

In this review we outline each propagation strategy and provide a summary of their relative advantages and disadvantages. In presenting case studies we draw heavily upon the research that the authors have performed in southern Costa Rica, as all three propagation methods have been evaluated in studies at the same research sites (Figure 8.1). These case studies have been conducted on land formerly used for cattle grazing or for growing coffee for at least 18 years. Sites are in the tropical premontane rain-forest zone, range in elevation from 1000–1500 m above sea level, and receive a mean annual rainfall of about 3500–4000 mm with a dry season from December to March.

8.1. Establishing tree seedlings from seed in nurseries

Establishing tree seedlings in nurseries from seed is by far the most common strategy used to propagate trees for restoration. Studies have either focused on establishing a broad range of species to create a baseline forest community (e.g.

Figure 8.1.
Location of the experimental sites used to test tree-propagation methods in Costa Rica

Butterfield, 1995; Parrotta and Knowles, 2001; Rodrigues *et al.,* 2009) or on using a few species as nurse trees to create infrastructure at a restoration site and accelerate the natural process of forest recovery (e.g. Holl *et al.,* 2011). Researchers typically either work with a tree nursery to obtain large numbers of seedlings (Carpenter, Nichols and Sandi, 2004; Holl *et al.*, 2011), harvest their own seeds from an adjacent forest and germinate and establish them in shade houses (e.g. Butterfield, 1995, 1996; Elliott *et al.*, 2003; Carpenter, Nichols and Sandi, 2004) or transplant seedlings that have established in a natural setting (often referred to as "wildlings") directly into a restoration site (Parrotta and Knowles, 2001). The diversity of species in a nursery is often limited to a few native and exotic commercially viable species, although the range is increasing in some regions as restoration efforts become more widespread. For example, 60–80 species are often available in nurseries in southeastern Brazil, where there are extensive restoration projects in the Atlantic forest region (Rodrigues *et al.*, 2009).

Establishing seedlings in nurseries from seed can yield large numbers of individuals. Seedlings are typically transplanted when they reach 20–40 cm in height (Carpenter, Nichols and Sandi, 2004; Holl *et al.*, 2011). In former pasture lands that are dominated by aggressive African pasture grasses, surrounding above-ground vegetation should be cleared periodically for 2–3 years to reduce competition and shading (Butterfield, 1995; Holl *et al.*, 2011); if this is not done (and sometimes even if it is), seedling mortality can be very high. When a full canopy cover is obtained, maintenance is no longer necessary as competition from pasture grasses and other ruderal vegetation is decreased as a result of shading.

Establishment of stands with either pure or mixed tree species has been broadly demonstrated to be successful in the tropics (e.g. Butterfield, 1995; Haggar, Wightman and Fisher, 1997; Lamb, 1998; Montagnini, 2001; Parrotta and Knowles, 2001; Calvo-Alvarado, Arias Richter, 2007; Butler, Montagnini and Arroyo, 2008). Survival and growth of different species can vary widely, however, and it appears that some are more able than others to tolerate the stressful microclimate and nutrient conditions found in degraded tropical landscapes (Butterfield, 1995; Parrotta and Knowles, 2001; Carpenter, Nichols and Sandi, 2004). The successful establishment of a given species can be highly site-specific (Butterfield, 1996). A number of authors have suggested that some large-seeded and shade-tolerant species are better introduced at later stages in succession, once an overstory canopy cover has developed and conditions are more favourable to seedling establishment (Parrotta and Knowles, 2001; Martínez-Garza and Howe, 2003; Cole *et al.*, 2011).

Lack of genetic variability among seedlings is a concern in many tropical forest restoration projects. Due to logistical constraints, most nursery endeavours (commercial and non-commercial) have often harvested seed from fewer than ten mother trees (Butterfield, 1995; Carpenter, Nichols and Sandi, 2004), and this can have strong ramifications for the long-term fitness of populations (Carpenter *et al.*, 1995). Once a canopy cover is established, recruitment of naturally dispersed species can be quite high, resulting in rapid forest recovery (Jones *et al.*, 2004; Butler, Montagnini and Arroyo, 2008), although the community composition of species can vary widely depending upon the nurse species planted (Parrotta and Knowles, 2001; Carnevale and Montagnini, 2002) and the availability of local propagules (Holl, 2007).

Case study

Holl *et al.* (2011) established a long-term restoration study spread across 100 km^2 in southern Costa Rica in 2004–2006. The 14 study sites are located between the Las Cruces Biological Station (8°47′7″ N; 82°57′32″ W) and the town of Agua Buena (8°44′42″ N; 82°56′53″ W). Each site incorporates three 50 × 50 m treatment plots – two active restoration plots and one passive or control restoration plot. Seedlings of four tree species (*Terminalia amazonia* (J.F. Gmel.) Exell, *Vochysia guatemalensis* Donn. Sm., *Erythrina poeppigiana* (Walp.) Skeels and *Inga edulis* Mart.) were established in two planting designs at each site – a plantation-style planting where the entire 50 × 50 m area was planted, and an island planting where trees were planted in six different-sized patches within the 50 × 50 m plot. The four species chosen are characterized by high regional survival, rapid growth and extensive canopy development (Nichols *et al.*, 2001; Carpenter, Nichols and Sandi, 2004; Calvo-Alvarado, Arias and Richter, 2007). *Terminalia amazonia* and *V. guatemalensis* are native timber species that produce valuable timber and favour establishment of native woody species in their understory (Cusack and Montagnini, 2004). *Erythrina poeppigiana* and *Inga edulis* are naturalized, fast-growing nitrogen-fixing species. Both are widely used in agricultural intercropping systems to provide shade and increase soil nutrients, and have extensive branching architecture; *I. edulis* also produces fruit that can attract birds (Pennington and Fernandes, 1998; Nichols *et al.*, 2001; Jones *et al.*, 2004). All four species were purchased from a local nursery.

All sites were cleared of above-ground vegetation with machetes prior to planting. Seedlings averaged 20–30 cm in height when planted. Ruderal vegetation was cleared every two to three months at all sites for 2.5 years. Establishment of planted seedlings was highly successful (>90 percent) and some sites reached canopy closure within two to three years (Holl *et al.*, 2011). However, growth rates were highly variable among sites and some have yet to develop a fully closed canopy even six years into the study. The reasons behind this disparity are unclear, however, but are probably related to prior land use. Seed dispersal and tree recruitment at this stage in the study (six years after planting) is largely comprised of early-successional species with few mid- to late-successional species (N = 55 species by 2010 survey; Cole, Holl and

Zahawi, 2010; Zahawi *et al.*, 2013). Overall, the strategy of planting a few widely available and hardy nurse species to accelerate natural forest recovery has been highly successful. However, the broad variation in both establishment and growth of planted seedlings, as well as the huge variation in seed dispersal and subsequent seedling establishment among sites, strongly underscore the importance of broadly replicating restoration studies across the landscape to avoid reaching erroneous conclusions based on a few sites.

8.2. Establishment by vegetative propagation

Vegetative propagation has been an integral technique for the establishment of trees in tropical agriculture, especially in the humid tropics, for many decades. A few commercial species of trees are also propagated vegetatively, such as Pochote (*Pachira quinata* (Jacq.) W. S. Alverson; also known as *Bombacopsis quinatum* (Jacq.) Dugand) (Hunter, 1987), beechwood (*Gmelina arborea* Roxb.) (Romero, 2004) and teak (*Tectona grandis* Linn. f.) (Husen and Pal, 2007). Whereas vegetative propagation has been used extensively in tropical agriculture and silviculture, it has received relatively little attention as a method for tree propagation in tropical restoration thus far (but see Perino, 1979; Ray and Brown, 1995; Chapman and Chapman, 1999; Granzow de la Cerda and Garth, 1999; Zahawi, 2005).

There are two main forms of vegetative propagation: (1) cuttings, which are typically 20–40 cm long, taken from young branches or shoots of trees; and (2) stakes, which are typically 2–2.5 m long, taken from branches that are pollarded from trees or extant live fence rows.

The establishment of trees from cuttings has several advantages, including ease of transport, availability in considerable quantities (once a mother tree is located, a considerable number of cuttings can be harvested), speed of planting (particularly if planted directly into the restoration site) and cost effectiveness. The application of cuttings for tree propagation in restoration activities has been limited to a few studies that have focused on the methodology (e.g. Ray and Brown, 1995; Itoh *et al.*, 2002; Bonfil-Sanders, Mendoza-Hernandez and Ulloa-Nieto, 2007), but cuttings have been widely used for enrichment planting of dipterocarp forests in Indonesia (Kettle, 2010). Most studies have reported mixed success, with high failure rates of a number of species despite the application of rooting hormones. A few studies have evaluated the possibility of using cuttings to propagate rare or endangered species (Danthu, Ramaroson and Rambeloarisoa, 2008; Ratnamhin, Elliott and Wangpakapattanawong, 2011), with some success with some species.

Itoh *et al.* (2002) found that rooting ability of 100 tropical trees in Malaysia was related to plant family and the growth characteristics of the species; fast-growing species that were generally of smaller mature stature typically rooted more readily. The ability to establish is also related to the type of cutting used; mature branches (harvested further down a stem) establish more readily than apical cuttings (Dick *et al.*, 1998; Danthu *et al.*, 2002) and leafy cuttings appear more successful at rooting than leafless cuttings (Brennan and Mudge, 1998; Dick *et al.*, 1998). Seasonality of timing when cuttings are harvested can also influence establishment success (Danthu, Ramaroson and Rambeloarisoa, 2008).

In contrast to cuttings, stakes have been widely used in agricultural practice throughout southern Mexico and Central America, especially in the humid tropics. Although the predominant use of the technique has been to establish live fences, stakes have also been used as host plants for agricultural crops such as vanilla and black pepper, and in some instances for erosion control (Perino, 1979; Sauer, 1979; Budowski, 1987; Budowski and Russo, 1993). In addition to these functions, trees often provide other benefits such as nitrogen fixation to improve soil quality, shade for coffee, fodder for cattle and firewood (Budowski and Russo, 1997; Martínez-Betancourt, Ramírez-Molinet and Rodríguez-Durán, 2000; Harvey *et al.*, 2005). Such species are also widespread throughout the landscape and have a proven track record of being hardy, withstanding not only the harsh conditions found in pastures but also tolerating extensive and repeated pollarding and other agricultural practices.

Stakes are typically planted as 2–3-m-tall branches ranging in diameter from 4 to 12 cm inserted directly into a planting site at a depth of 20–30 cm (Budowski and Russo, 1993; Martínez-Betancourt, Ramírez-Molinet and Rodríguez-Durán, 2000; Zahawi, 2005), although stakes at tall as 4–4.5 m can also be established readily (Zahawi, 2008). Accordingly, some degree of above-ground vertical stratification can be created at the time of planting. Establishment success appears to vary widely and is dependent on a number of variables, including geographic location, elevation, rainfall and planting season (Budowski and Russo, 1993; Alonso *et al.*, 2001; Zahawi, 2005). Initial stake size (both height and diameter) affects survival and growth and can also have an impact on biomass production rates (da Costa *et al.*, 2004; Zahawi, 2005; 2008; Zahawi and Holl, 2009). Stakes also develop greater above- and below-ground biomass than seedlings in the initial years after planting; below-ground architecture is also distinctly different, with stakes producing extensive lateral roots while lacking a centralized taproot (Zahawi and Holl, 2009). Whereas most farmers consider it important to plant stakes just after a full moon (Budowski and Russo, 1993), the effect of moon phase has been examined empirically in only one study; only slight differences were found in a few growth indicators but not for survival (Alonso *et al.*, 2002).

Although stakes have long been used in agricultural practice and there is often widespread local knowledge of how to establish the species that are utilized in a given location, much of the information on species establishment, such as timing and seasonality of planting, has not been published or quantified experimentally (but see Alonso *et al.*, 2001; Zahawi, 2005). This information is traded among practitioners and stakeholders, and has occasionally been compiled in anecdotal form (e.g. Sauer, 1979; Budowski and Russo, 1993). Although the literature documents several hundred species that can establish vegetatively (Budowski and Russo, 1993; Martínez-Betancourt, Ramírez-Molinet and Rodríguez-Durán, 2000; Harvey *et al.*, 2005), farmers overwhelmingly rely on only a few species with widespread distribution and use; however, species choice does vary regionally and by country. Farmers' species selection is focused naturally on features that are important to them, e.g. species that are not toxic to livestock, can hold barbed wire and are able to withstand regular pollarding (Sauer, 1979; Budowski and Russo, 1993). In contrast, restoration ecologists would likely focus on species with fruit that attract frugivores, an ability to shade out pasture grasses, extensive canopy architecture and rapid growth rates. Accordingly, studies are needed to better document the establishment needs and abilities of species of interest to restoration. In addition, an evaluation of the functional traits shared among species that establish vegetatively would be particularly useful and would facilitate the search for potential forest species that could be of value to restoration.

Case study

Plots were established at three field sites in Costa Rica to evaluate growth and survival of stakes and compare their performance with standard nursery-raised seedlings (Zahawi and Holl, 2009). At each site, stakes were harvested from 20–30 individual fence trees of each of ten species from nearby live fence rows (less than 3 km away from the trial site) and planted vegetatively in rows at 1.5 m intervals. Species were chosen based on their common use as live fence rows in the area. Stakes were approximately 2 m tall at planting. A pointed pole was inserted into the ground to open a 15–20-cm-deep hole. The stake was then inserted in the hole and soil was lightly compacted around its base. All stakes were planted in July (wet season) and were monitored for three years for survival and above-ground development.

Survival differed between species, ranging from more than 90 percent to less than 30 percent;

survival of some species was highly site-specific. For most species, stakes with greater initial diameter had a greater probability of survival. Species varied enormously in above-ground biomass development, and canopy cover ranged from less than 2 m^2 to more than 10 m^2 in the third year. Variability between sites was high. Not surprisingly, sites where survival and growth of stakes were high were the same sites where establishment rates for planted seedlings were high (Holl *et al.*, 2011). In comparing planting strategies between the two studies, three-year-old *Erythrina poeppigiana* stakes had greater canopy cover than saplings of the same age, although their height was similar. Several species established from stakes produced fruit in the second and third year after planting. This is not surprising given that stakes are pollarded from reproductive adult trees, conferring an advantage over planting seedlings that can take decades to produce fruit and attract seed dispersers.

8.3. Direct seeding

Direct seeding is by far the cheapest way of reintroducing vegetation (Lamb, Erskine and Parrotta, 2005; Cole *et al.*, 2011), but tree seeds can be hard to acquire and the rate of success is highly variable. Seeds are typically harvested from trees or the forest floor in nearby forests (Doust, Erskine and Lamb, 2006; Sampaio, Holl and Scariot, 2007; Cole *et al.*, 2011), although in some cases they can be purchased (Engel and Parrotta, 2001). Seeds are either placed on the soil surface or buried. Many tropical forest tree seeds are recalcitrant (i.e. they rapidly lose viability when dried), making storage impossible, and the technique of bulking up seed in the greenhouse or field plots, which is commonly used for temperate tree species, is not feasible for tropical forest trees.

As with the afore-mentioned propagation methods, genetic variability of seed stock is often low. Collecting tropical seed from a wide variety of species can be difficult; many tropical forest trees do not set seed every year and individuals of a given species are often widely dispersed. As a result, it is not uncommon to harvest seeds from fewer than five mother trees and in some cases only two or three (Doust, Erskine and Lamb, 2006; Sampaio, Holl and Scariot, 2007; Garcia-Orth and Martinez-Ramos, 2008; Cole *et al.*, 2011). This can have strong effects on germination and survival, depending upon the quality of the seed source, and could result in reduced genetic diversity in future generations.

Direct seeding can be applied in two principal ways: (1) at the onset of the recovery process of the site; and (2) at later stages in the recovery process, typically after a canopy cover has formed.

Direct seeding at the initial stages of recovery has been tested in several small-scale experimental studies, but has not been considered a viable restoration option at a large scale because of the high rate of failure and the challenge of acquiring and storing sufficient seed (Ray and Brown, 1995; Engel and Parrotta, 2001; Woods and Elliott, 2004; Doust, Erskine and Lamb, 2006; Sampaio, Holl and Scariot, 2007). In most cases some establishment occurs, but the variability across species and sites is highly unpredictable. Seeds and recently germinated seedlings typically succumb to a host of setbacks, including pathogen attack, predation and desiccation (Augspurger, 1984; Nepstad, Uhl and Serrao, 1990; Chapman and Chapman, 1999; Engel and Parrotta, 2001; Cole, 2009; Gallery, Moore and Dalling, 2010; Cole *et al.*, 2011). Small seedlings can also be difficult to see among ruderal vegetation and may be removed during routine vegetation clearing. Predators typically remove a larger proportion of smaller seed than larger seed, and larger-seeded species tend to have greater establishment success because they have larger amounts of stored resources (Camargo, Ferraz and Imakawa, 2002; Jones, Peterson and Haines, 2003; Doust, Erskine and Lamb, 2006; Vieira and Scariot, 2006a). In turn, burial appears to increase seed survival and germination compared with surface placement (Woods and Elliott, 2004; Doust, Erskine and Lamb, 2006; Garcia-Orth and Martinez-Ramos, 2008). Lastly, seasonal timing of planting can have a considerable effect on long-term survival, especially in areas with a prolonged dry season (Ray and Brown, 1995; Vieira and Scariot, 2006b).

Direct seeding seems to be most effective for larger-seeded and later-successional species that are introduced as part of enrichment planting after the canopy has closed (Nepstad, Uhl and Serrao, 1990; Hooper, Condit Legendre, 2002; Cole *et al.*, 2011). These species are often underrepresented in the initial stages of forest recovery because of their short-range dispersal. Studies comparing establishment at different successional stages indicate that, although germination rates are similar among stages, long-term survival is usually higher in sites that have tree canopy cover (Bonilla-Moheno and Holl, 2010; Cole *et al.*, 2011). In contrast, Camargo, Ferraz and Imakawa (2002) found higher survival of large-seeded species in open highly degraded sites and in pastures than in young and mature forest in lowland areas in Brazil.

To date, we know of no large-scale tropical forest restoration projects that introduced forest trees through direct seeding. However, some authors have suggested that seeding species with relatively high germination and survival rates at the early seedling stage should be a component of a mixed restoration strategy that includes seeding, planting seedlings and allowing for natural regeneration of different species depending on their life history (Cabin *et al.*, 2002; Sampaio, Holl and Scariot, 2007; Bonilla-Moheno and Holl, 2010). In turn, larger-seeded, later-successional species may be introduced in small patches in forests with an overstorey, as introducing such species over large areas is probably not feasible because of lack of seeds.

Case study

In our study area in southern Costa Rica, we evaluated the ability to establish from direct seeding of six mid- to late-successional tree species in three distinct habitats: recently abandoned pasture, young plantation (approximately three years old) as described earlier and young secondary forest (approximately eight years old). The direct seeding study was replicated across four research sites (Cole *et al.*, 2011). Species were sown to an average depth of 3 cm, and germination and survival were monitored for two years. Germination rates after two years ranged from near complete failure in one species to 26–31 percent for four species and 94 percent in the sixth species. Overall germination was similar among the three habitats. However, survival was higher in plantations (75 percent) than in the other two habitats (~45 percent). Plantations also had greater overall biomass production at the end of the study, which appeared to be due to higher nitrogen availability as two of the four plantation trees were nitrogen-fixing species. Results indicate that direct seeding of later-successional species into young, recovering habitats with some degree of overstorey cover can circumvent their dispersal limitation and contribute to higher species diversity in the forest.

8.4. Choosing an appropriate restoration strategy

A first stage in any restoration project is to clearly identify the goals. These goals and specific objectives will necessarily need to be developed along with a consideration of the resources (e.g. financial, labour, sources of seeds or seedlings) available to achieve these goals and the natural resilience of the target ecosystem (Holl and Aide, 2011). A goal of most tropical forest restoration projects will be to restore the species composition and processes of the forest before it was disturbed. However, given the competing needs of providing for human livelihoods and maximizing certain ecosystem services, there will be trade-offs concerning which goals will be prioritized, such as species diversity, carbon sequestration, erosion control, or providing wood or food products used by humans.

The degree of passive recovery of degraded tropical lands is highly variable, depending on the ecology of the system, land-use history and the surrounding landscape mosaic (reviewed in Holl, 2007). Therefore, a critical first step in a restoration project is to determine which species will resprout or colonize naturally and, therefore, may not need to be introduced (Holl and Aide, 2011). Second, it is a wise investment of resources to conduct smaller field trials prior to planting large-scale projects, as propagation methods and species differ in their success rates from one location to another. For projects that span large regions, it is important to conduct pilot studies at multiple sites given the high variability in success over even relatively small spatial scales (Butterfield, 1996; Zahawi and Holl, 2009; Holl *et al.*, 2011) as a result of numerous factors, including land-use history, soil physical and chemical properties, soil microbial communities, competition with existing vegetation and differences in microclimates. All these recommendations take time and money to implement, but in the long run will help to ensure the most efficient allocation of restoration resources and will minimize the risk of large-scale restoration failure.

Selecting an appropriate tree introduction method requires knowledge of the natural history of the species available. While this information is lacking for many species, the number of studies screening germination rates (e.g. Sautu *et al.*, 2006), seedling survival rates (e.g. Butterfield, 1995) and even cuttings (Itoh *et al.*, 2002) has increased in the past two decades. Species that have low seed germination rates, have complex germination triggers or produce small numbers of seeds are not well suited to direct-seeding efforts, given the large losses that typically occur as a result of predation, herbivory and pathogens in the field. Similarly, only certain species have known vegetative propagation abilities.

It is also important to consider how many species will be introduced. Many tropical restoration efforts plant a small number of tree species (often fewer than ten) to facilitate colonization and establishment of a typically highly diverse native flora and fauna, although in a few studies more tree species (20–30) have been planted to represent a range of growth rates and dispersal guilds (Lamb, 2011). It is much less common to plant more than 30 species, given the necessary knowledge for propagation and resources needed, although some restoration efforts strive for diverse plantings (e.g. 60–80 species; Rodrigues *et al.*, 2009) that include vines and shrubs. Finally, some species may not be commercially available so it is necessary to collect seed and then determine whether it is more efficient to introduce each species directly or establish them first as seedlings in a nursery.

The issue of introducing sufficient genetic variability into a restoration site is of concern in all the propagation methods discussed above. Forestry literature highlights the importance of using diverse genetic sources, as well as selecting for high-quality genotypes (reviewed in Carnus *et al.*, 2006; Kettle, 2010). While some restoration projects harvest material from many individuals, it is not uncommon, with all the propagation techniques described above, to harvest stock from only a few individuals, particularly when the number of source trees is limited. Studies examining the potential implications of such narrow selections (e.g. Carpenter *et al.*, 1995; Dick *et al.*, 1998) present compelling results, with high variability in the establishment and growth of individuals from different genetic stocks.

In many cases, the cost of different propagation methods will be an overriding consideration, given that most projects are financially constrained. For active restoration projects, direct seeding represents the most economical route and can be 20 to 30 times less expensive to carry out than traditional nursery plantings (Engel and Parrotta, 2001; Cole *et al.*, 2011). Cuttings represent a similar cost-effectiveness to direct seeding if they are directly planted out upon harvesting; however, this is often not the case. When established in nurseries, cuttings represent a similar cost investment to establishing from seed; accordingly, this method should only be applied to species that have demonstrated low seed fecundity, or species that are rare or endangered. The cost of using stakes is intermediate between direct seeding and using cuttings established in nurseries, with cost estimates ranging from two to ten times cheaper than nursery stock (Zahawi and Holl, 2009). Growing and planting seedlings is usually the most expensive strategy but it is also the most commonly used and the most widely tested methodology.

Logistical considerations, such as challenges of moving propagative material and the availability of propagation facilities, also affect species selection. Seeds and direct-harvested cuttings are the easiest propagules to transport. Stakes are not only cumbersome but care must be taken when transporting them so as not to damage the cortex, which can impair their establishment ability (Zahawi, personal observation). Accordingly, using stakes is appropriate only when vegetative material is available relatively close to a restoration site or the need to use this method outweighs the increased cost of transporting individuals of certain species. Seedlings are intermediate in terms of ease of transport but require shade-house facilities to propagate, which implies additional costs. Clearly, the relative costs and logistical considerations of different strategies will vary across restoration projects, depending on availability of seed and nursery facilities, transport distances for vegetative propagules and other local conditions.

Each of the three strategies has its own advantages and disadvantages, and it is likely that in most cases a combination of the different propagation methods is the best restoration approach. In addition, site-specific conditions, the surrounding landscape and other factors specific unique to a given restoration area will necessarily dictate the most appropriate strategy.

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Chapter 9

Seed availability for restoration

David J. Merritt and Kingsley W. Dixon

Kings Park and Botanic Garden, West Perth, Australia

The role that restoration plays in species conservation is increasingly recognized in global forums. For example, the recent Conference of the Parties to the Convention on Biological Diversity (COP-10) highlighted ecological restoration as a significant opportunity for achieving global conservation goals (CBD, 2010a). But some of the fundamental challenges to achieving global restoration targets, such as those set out in the Global Strategy for Plant Conservation 2011–2020 (CBD, 2010b), are in need of broader recognition. Contemporary restoration programmes aim to restore biodiverse plant communities. In practice this means the return of tens to hundreds of species in many ecosystems. Large-scale plant reintroductions (hundreds to tens of thousands of hectares) must be underpinned by the effective use of seeds of wild species. This in turn requires sufficient biological and technical knowledge of a large number of species to enable the collection, storage and germination of seeds and establishment of seedlings.

9.1. Landscape-scale restoration requires large quantities of seed

Options for the active return of plant species to degraded sites include direct seeding, planting of seedlings and the spreading of appropriately managed topsoil containing seeds (Koch, 2007; Rokich and Dixon, 2007). Each of these methods can be used exclusively or in combination, depending on the size of the restoration programme, the available physical and biological resources and the biological characteristics of the available plant material (e.g. the seed-storage characteristics). For all three options, seeds are fundamental, being spread to site through their incorporation into returned topsoil, broadcast by hand, machine planted (e.g. drill seeding or aerial seeding) or sown in a nursery for seedling production. Properly handled topsoil can be very effective at restoring plant communities (Koch, 2007; Rokich and Dixon, 2007). However, at most restoration sites seed-containing topsoil is limited or unavailable. For restoration at the landscape-scale, direct seeding is often the most viable means of initiating the return of biodiverse plant communities (Merritt and Dixon, 2011).

A reliable supply of seeds is critical to successful restoration. What is not always recognized are the constraints surrounding the quantity of seed required to achieve restoration goals and its availability (Merritt and Dixon, 2011). Insufficient, inconsistent and uncoordinated seed supply can be a significant limiting factor in restoration programmes. Even at the local or regional scale, factors such as the availability of seeds, the technical knowledge, training and licensing of the seed collectors, the cost of seeds, and the biological and technical knowledge necessary to correctly process, store, break dormancy and deliver seeds to restoration sites contribute to seed-supply shortfalls. At the landscape scale, these factors can be greatly compounded by the very large quantities of seeds needed for restoration.

Many restoration programmes are planned or underway across the globe, aimed at restoring thousands or even tens of thousands of hectares, often in poorly studied ecosystems with little available information on seed attributes or restoration technology. With current restoration technologies the amount of seed required for such programmes can be calculated to be in the hundreds of tonnes, far exceeding the seed-collecting capacities of government agencies, non-governmental organizations (NGOs) and commercial operations, as well as the available seed resource that can be practically and ethically collected from wild plant populations. Seed availability is thus one of the most significant challenges to large-scale restoration programmes (Broadhurst *et al.*, 2008; Rodrigues, Lima *et al.*, 2009; Gibson-Roy *et al.*, 2010; Merritt and Dixon, 2011; Tischew *et al.*, 2011).

There are many examples of the quantities and costs of seeds required for landscape-scale restoration. In the agricultural zone of southwest Western Australia, a 14 million ha agricultural zone within a Mediterranean-climate, biodiversity hotspot, over 93 percent of the landscape has been cleared of native vegetation over the past 60 years, resulting in numerous sustainability and productivity problems, including dryland salinity, soil erosion and weed invasion (Prober and Smith, 2009). To combat landscape salinization an estimated 20-70 percent of the landscape would need to be returned to deep-rooted, woody perennial vegetation (Prober and Smith, 2009). Using a conservative seeding rate of just 1.5 kg/ha (Jonson, 2010), attempting to restore plant communities to just 20 percent of this landscape would require 4200 tonnes of seeds. In tropical forests in Borneo, restoration projects plant between 500 and 2500 seedlings/ha (Kettle *et al.*, 2011). Even at a planting density of just 500 seedlings/ha, over 7 billion seedlings would be required to restore the estimated 14.3 million ha of degraded forest (Kettle *et al.*, 2011). In the United States, the Bureau of Land Management (BLM) purchased 125 tonnes of seed of forb species in one year for the Great Basin Restoration Initiative (Shaw *et al.*, 2005) and in 2007 the BLM spent US$50 million on seeding grass species in the Great Basin, Mojave and Sonoran Deserts (Knutson *et al.*, 2009). On a similar scale, the cost of seed purchases for restoration of 20 000 ha of land disturbed through mining activity in the semi-arid Pilbara grasslands of Western Australia has been estimated to exceed AUS$100 million at current prices for wild-collected seeds (Merritt and Dixon, 2011).

9.2. Seeding rates necessary to delivery restoration outcomes

The quantity of seed required to ensure an acceptable level of seedling establishment can vary substantially across different biomes. Ideally, seeding rates are based on known parameters and data, including seed size, viability, germination and establishment rate (Gibson-Roy *et al.*, 2010). These parameters of seed quality are captured in the concept of "pure live seed" (a measure of the purity, viability and germination capacity of a seed batch), an accreditation tool used for evaluating seeds produced via commercial farming of wild species in the United States and some parts of Europe (Jones and Young, 2005). If information on seed quality is not gathered prior to seeding, it is not possible to determine the success (or otherwise) of direct seeding through monitoring and documentation of seedling emergence to determine the proportion of seeds that emerge. In restoration practice, seed-quality analysis prior to seeding, and monitoring of the results following seeding, is often not done. Commonly there is little published information available to guide setting of seeding rates for local projects, or criteria to evaluate success, reducing the incentive for practitioners to strive for improvements in seed-use efficiency. Many studies of direct seeding are done on a very small scale (e.g. a few square metres) to investigate the effects of seed addition and/or seed treatments on seedling emergence and establishment. These studies do not always report seeding rates on a weight/area basis (e.g. kg/ha), but rather the addition of a defined number of seeds into a small plot or simply employ an unknown number of seeds. However, some examples of seeding rates for different biomes are available that can be used to substantiate the amount of seed required for restoration (Table 9.1).

9.3. Constraints to seed supply for landscape-scale restoration

In most restoration projects the majority of seeds are collected from wild plant populations. This presents some challenges, given that many wild populations, particularly those surrounding agricultural, pastoral and urban lands, are highly fragmented, often degraded and under stress. The amount of seed available to collect from wild sources can fluctuate significantly from year to year because of such factors as the environmental conditions experienced by the maternal plants, pollen flow, a requirement for disturbances (e.g. fire) that promote mass flowering and fruiting of some species and species biology (Jones and Young, 2005). Relying solely on seeds collected from the wild will increasingly result in supply shortfalls as the demand for seeds increases to match the scale of restoration.

The timing of seed collection is critical to a successful outcome as there is often a window of only a few days or weeks between when seeds are ready to collect and when they are dispersed and no longer available for collection. It is important to consider the phenology of seed development, particularly the timing of seed maturation, to ensure that the collected seeds are suitably resilient to post-harvest handling. Seeds should be collected as near as possible to the point of natural dispersal to ensure that quality, desiccation tolerance (for orthodox seeds) and longevity are maximized (Hay and Smith, 2003). Kodym, Turner and Delpratt (2010) demonstrated, for example, that several species of *Lepidosperma* (Cyperaceae), which contribute important understorey components of temperate Australian woodland, shed viable seeds quickly but retain non-viable seeds for some months. Further, in these species viable and non-viable seeds look similar, meaning that incorrect timing of collection (i.e. too late) could result in only non-viable seeds being collected, while giving the impression that viable seeds are available (Kodym, Turner and Delpratt, 2010). The importance of collection timing has also been recently highlighted for tropical species. The reproductive ecology of important trees species, including dipterocarps, presents

TABLE 9.1.
Examples of seeding rates used in restoration programmes in different biomes

Region	Biome	Seeding rate (kg seed/ha)	Source
Australia	Mediterranean woodland	1.5	Jonson (2010)
Australia	Arid grassland	5–7	Merritt and Dixon (2011)
Australia	Temperate grassland	50–110	Gibson-Roy *et al.* (2010)
Germany	Semi-natural grassland	20–100	Baasch, Kirmer and Tischew (2012); Kirmer, Baasch and Tischew (2012)
Northwestern Europe	Ex-arable grassland	10–100	Kiehl *et al.* (2010)
Northwestern Europe	Grassland	20–40	Török *et al.* (2011)
United Kingdom	Calcareous grassland	1–40	Stevenson, Bullock and Ward (1995)
United States	Continental sagebrush	2–8	Williams *et al.* (2002)

particular challenges for large-scale seed supply (Kettle, 2010). Dipterocarp seed production is sporadic and unpredictable, with mass flowering and fruiting events (seed masting) being a common but infrequent occurrence (Kettle, 2010; Kettle, 2011). The window for seed collection is short, usually a few weeks, and seed-masting events are separated by years of low seed production (Kettle, 2011). Many tropical forest species produce recalcitrant seeds (Sacande, 2004), including many of the species important for timber. Recalcitrant seeds do not survive desiccation and cannot be stored for more than a few weeks or months (Berjak and Pammenter, 2008). Storage behaviour of recalcitrant seed means that it is not possible to take advantage of seed-masting events through the collection and storage of seeds for use in years of low production. Recalcitrant seeds must be germinated immediately and the seedlings held in a nursery for planting into restoration sites (Kettle, 2010; Kettle, 2011).

A need to source local provenance seeds for restoration can also create challenges. Seed of local provenance is best defined as seed that is genetically representative of a species growing within a particular climate, habitat, soil type and profile in the landscape. Seed provenance is important to restoration as local genotypes are assumed to be better adapted to local environmental conditions and, therefore, more likely to establish (Krauss and Koch, 2004; McKay *et al.*, 2005; Bischoff, Steinger and Müller-Schärer, 2010; Jonson, 2010; Mijnsbrugge, Bischoff and Smith, 2010). Sourcing seeds of local provenance can be problematic, particularly in highly fragmented landscapes where small, remnant patches of vegetation are separated by large areas of land cleared for agriculture, infrastructure and residential development. In these localized areas the demand for seeds can easily exceed the supply and there may be some risks of detrimental effects on the viability of the source vegetation caused by overharvesting of seeds (Broadhurst *et al.*, 2008).

9.4. Approaches to improving seed availability for restoration

Developing appropriate seed-banking procedures

Seed banking is a crucial link in the restoration chain. Correct handling and storage allows orthodox seeds to be banked over many seasons and allows practitioners to capitalize on high-seeding years, providing a resource for large restoration projects. Careful control of the storage environment will ensure that seed viability is maintained. Flexibility in the available storage conditions is preferable, and seeds should be stored under conditions appropriate to their storage behaviour, dormancy type and designated storage duration (Merritt and Dixon, 2011). Recognizing that all seeds go through a storage phase prior to use in restoration and putting in place the intellectual and infrastructural capital required to curate the seeds appropriately will ensure that the quality of the seed resource is maintained. At present seeds for use in restoration are stored almost exclusively by end users, including the commercial seed industry, mining companies, NGOs and community-based groups. As a result, storage facilities holding seeds for restoration are commonly low on technology, have limited access to knowledge and training in modern seed science, have little or no capacity for problem solving or research and, in the case of the commercial seed merchant, are profit-driven, meaning only those plant species that are profitable (i.e. those producing seeds that are easily accessible, robust to the storage conditions and more reliable at the establishment phase) will be sought, traded and employed in restoration. Inadequate resourcing of restoration seed banks is a rapidly emerging bottleneck hampering landscape-scale restoration. Restoration seed banks must be developed by adapting principles and technologies put in place for seed banks conserving biodiversity and food crops, with the crucial difference that the volume of seed required to address biodiverse landscape-scale restoration compels restoration seed banks to store hundreds of tonnes of seeds (Merritt and Dixon, 2011).

Improving seedling establishment

A major limitation to the effectiveness of direct seeding is the poor conversion of seeds into established seedlings (James, Svejcar and Rinella, 2011; Merritt and Dixon, 2011). Failed seedling establishment is a significant contributing factor to the huge quantities of seeds required for restoration and the inability to re-establish functional plant communities. Across a range of habitats, commonly less than 10 percent (and often as low as 3 percent) of seeds delivered to site germinate and establish (Merritt and Dixon, 2011). In Mediterranean southwest Australia, emergence rates of 1–17 percent have been reported for a range of *Banksia* woodland native species (Turner *et al.*, 2006; Rokich and Dixon, 2007). Similarly, in the arid grasslands of the United States, 7–17 percent establishment of germinated seeds was recorded for three grasses, and modelling of seed fates across four restoration sites calculated the probability of a seed producing an established seedling to be less than 0.06 (James, Svejcar and Rinella, 2011). Low seedling establishment is also reported for tropical forests. A restoration trial using three mature forest species to seed land previously used for slash-and-burn agriculture in Mexico's Yucatan Peninsula found on average that 5–41 percent of seeds germinated and emerged, and that 3–35 percent of these seedlings established (Bonilla-Moheno and Holl, 2010). In central Amazonia, seedling emergence of 12–33 percent has been reported across 11 native tree species seeded into abandoned pasture lands (Camargo, Ferraz and Imakawa, 2002). Seed losses accrue not just through failed germination and establishment, but also through wind and water erosion and predation (Holl *et al.*, 2000; Doust, 2011).

Research and technological development is needed to reduce the wastage of seeds during delivery and establishment. Seed-enhancement treatments must be explored to increase seed germination performance and seedling establishment. Seed-enhancement treatments include priming, coating and pelleting. Much of this technology is routinely applied through the agricultural and horticultural biotechnology industries, but as yet has not been widely adopted in the native seed industry. However, priming has been demonstrated to increase seedling emergence of native grass species under field conditions (Hardegree and Van Vactor, 2000), and simple techniques of on-farm seed priming are used for cereals and legumes to improve crop establishment (Harris *et al.*, 1999). Seed pelleting has been demonstrated to increase seedling emergence of *Banksia* woodland species in southwest Western Australia, as well as decreasing predation and losses through wind erosion (Turner *et al.*, 2006). Seedling establishment rates can also be improved by correctly timing seed delivery to site and employing simple treatments such as incorporation of seeds into the soil (Turner *et al.*, 2006).

Increasing seed supply

In variously termed seed orchards, seed farming or seed-production areas, growing wild plant species specifically to harvest their seeds for restoration is receiving increasing attention as a part of the solution to seed-supply shortfalls. Options for seed-production areas include the setting aside of wild populations of plants for dedicated seed collection, the growing of plants in pots under nursery conditions for annual harvesting of seeds (Koch, 2007; Gibson-Roy *et al.*, 2010) or the development of purpose-designed broadacre seed farms where plants are grown using agricultural cultivation and harvesting techniques (Shaw *et al.*, 2005). Some common challenges to developing viable seed-production enterprises for a wide range of species include a limited knowledge of seed-propagation and plant-husbandry requirements, and the need for rigorous seed certification and quality-control procedures and to effectively manage genetic considerations, including the potential provenance variation of source-plant material and the genetic consequences of seed production (Gibson-Roy *et al.*, 2010; Tischew *et al.*, 2011). Other issues relate to the inadvertent selection processes inevitably introduced via source-plant seed collection, maternal-plant growth and survival and harvesting techniques (Mijnsbrugge, Bischoff and Smith, 2010). Nevertheless, programmes of research, development and commercial supply through large-scale, certified wild-seed production are in place for large-scale restoration programmes such as those under the Great Basin Restoration Initiative of the United States[11] (Shaw *et al.*, 2005). On a similarly large scale, the SALVERE Project,[12] across central Europe, includes research into the seed-production potential of semi-natural grasslands as a source of seeds for restoration. The Millennium Seed Bank's UK Native Seed Hub Project[13] aims to establish seed production for lowland meadows and semi-natural grassland across the United Kingdom in partnership with the commercial and restoration sectors. At a more regional scale, the potential for NGOs and local communities to develop and manage seed production areas to increase the supply of understorey species has been successfully demonstrated for the Grassy Groundcover Restoration Project across south-eastern Australia (Gibson-Roy *et al.*, 2010). This project produced 92 kg of seeds of approximately 200 native herbaceous species over two years for the restoration of ex-agricultural land (Gibson-Roy *et al.*, 2010).

9.5. Conclusion

Seeds are fundamental to large-scale restoration, being the only viable means of reintroducing plants at the 100–1000 km^2 scale. But obtaining seeds of wild species is a significant challenge to landscape-scale restoration. Key areas of seed biology and technology underpin restoration, and optimizing each step in the chain of seed usage in restoration, from collection to delivery to site, is necessary. Knowledge of these key areas is complex when dealing with biodiverse plant communities and species-specific information. Seed-enhancement techniques for each species must be tailored to site-specific needs for effective restoration. This includes consideration of abiotic factors such as the landform stability, slope, aspect and the available growing medium (soil conditions which are often heavily different to those prior to disturbance) and hydrological aspects, including the reliability and seasonality of rainfall and soil-moisture retention properties. The unification of science-based seed knowledge with the infrastructure to support large-scale seed management and the development of effective working relationships between seed scientists, restoration practitioners, the commercial seed industry and the local community will ensure seeds are used to their full potential.

[11] http://www.blm.gov/id/st/en/prog/gbri/technology/native_plants.html

[12] http://www.salvereproject.eu

[13] http://www.kew.org/news/uk-seed-hub.htm

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Insight 6

Seed availability: a case study

Paul P. Smith

Millennium Seed Bank, Royal Botanic Gardens, Kew, United Kingdom

A major constraint for afforestation and restoration programmes around the world is the lack of availability of large numbers of high-quality seeds of indigenous species with suitable provenance and accompanying data.

As part of an ongoing seed-longevity study, Kew's Millennium Seed Bank (MSB) recently identified a range of tree species listed on the World Agroforestry Centre's Tree Seed Supplier Directory (TSSD) that were not present in the MSB's collections and which would be suitable for the study.[14]

Seed availability

Kew targeted 30 of the largest public-sector and commercial seed suppliers on the TSSD who between them should, according to the Directory, have been able to supply 1624 species meeting Kew's requirements. However, of the 30 suppliers listed, seven could not be contacted and one was on the list twice. The remaining 22 were contacted, but only seven responded, representing a 24 percent success rate for supplier responses.

Once contact had been made, Kew requested a total of 633 species listed as available from the seven suppliers. A minimum number of 2000 seeds were requested, and minimal accompanying data on seed origin and storage conditions were specified.

Eventually, six months after the process begun, Kew was able to secure collections of 218 unique species. This represents an overall seed-supply success rate of 13 percent of the total number of species theoretically available and 34 percent of the species advertised by the seven suppliers who were successfully contacted.

Data quality

A subset of 572 species on the TSSD were checked for current name status, and it was found that 48 percent of the names on the list are no longer valid (i.e. they are synonyms). When the correct names were compared with the MSB's accession list it was found that 25 percent of the collections were already in the MSB.

The following provenance data accompanied the collections received: wild/cultivated origin (48 percent of collections); date of collection (88 percent); country of origin (100 percent); region of origin (65 percent); precise locality (14 percent).

Seed quality

Seed-quality testing is currently taking place. However, the collections were accompanied by the following information on seed processing: drying conditions (specified for only 15 percent of collections); date of storage (89 percent); relative humidity during storage (15 percent); and temperature of storage (100 percent).

Conclusion

All of the above indicates the common difficulties encountered in sourcing high-quality seed collections in reasonable numbers and with minimal provenance data, even from reputable sources.

[14] See http://www.worldagroforestrycentre.org/Sites-old/TreeDBS/tssd/treessd.htm

Insight 7

The role of seed banks in habitat restoration

Paul P. Smith

Millennium Seed Bank, Royal Botanic Gardens, Kew, United Kingdom

Seed banks have a major role to play in habitat restoration, both as a source of material and in solving research problems related to reintroducing species back into the landscape (Hardwick *et al.*, 2011; Smith *et al.*, 2011).

Seed banks have an important advantage over nurseries in that they can store a large amount of genetic diversity in a very small space. For example, a typical 30 m^3 cold room in Kew's Millennium Seed Bank stores 20 000 seed collections totalling 1 billion seeds. In addition, seeds kept under cool, dry conditions are more secure than seedlings in a nursery, the latter being more susceptible to pests and diseases, extreme weather etc. From this perspective, it makes sense to store plant diversity as seed right up to the time when it is needed. Finally, from the restoration practitioner's viewpoint, direct seeding is far more cost-effective than reintroducing seedlings or saplings (see Section 8.4). However, for successful re-seeding, research is required to optimize germination and survival.

Seed-conservation research and expertise with direct relevance to restoration programmes includes seed sampling, collection, handling and developing appropriate storage methods (short, medium and long term). Seed morphology can also inform practitioners about natural dispersal mechanisms. However, perhaps the most important contribution that seed banks make is in developing optimal germination protocols, taking into account the physical and physiological dormancy mechanisms present in so many wild species. Most seed banks routinely carry out germination testing to test for viability. However, for wild species there are frequently challenges associated with dormancy mechanisms that need to be characterized, and appropriate pretreatments or priming methodologies developed (Probert, 2000; Merritt *et al.*, 2007).

Kew's Millennium Seed Bank (MSB) is currently the only global repository for wild species. It stores seeds from 141 countries, and every collection is tested for dormancy and germination. Optimal germination protocols and information on other traits, such as seed storage behaviour, are freely available through Kew's Seed Information Database on line.[15] The MSB's Seed Information Database currently contains information on more than 11 000 tree and shrub species.

For United Kingdom restoration practitioners, Kew has gone a step further and produced a germination predictor tool that takes into account where and when seeds are collected, and uses this information to predict optimal germination protocols.[16] This approach takes variation in local genotypes and climate into account.

Many national and regional seed banks fulfil similar roles locally. Seed banks with a strong restoration-ecology focus that provide both material and methodologies include Kings Park (Western Australia); Plant Bank (New South

[15] http://data.kew.org/sid/

[16] See http://www.kew.org/science-research-data/databases-publications/uk-germination-tool-box/

Wales, Australia); Chicago Botanic Garden's Plant Conservation Science Center (United States); China's Gene Bank of Wild Species in Kunming; and Kirstenbosch Botanical Garden in South Africa. In addition to these specialist institutes, many forestry gene banks support afforestation of native species. A recent survey of government tree seed centres in 12 African countries (Kew, unpublished), found that, collectively, these institutions supply 40 tonnes of seeds and 398 million seedlings of 558 species each year. The majority of seeds and seedlings supplied are of exotic species. However, all of the forestry institutions surveyed also supply indigenous tree seeds and seedlings, albeit in smaller amounts than exotics. In developed countries, capability related to the propagation and use of indigenous species is far more advanced. For example, each year Poland's State Forests supply 650 tonnes of seeds of native tree species from 92 different seed zones, which are used to produce around 850 million seedlings for introduction in to the landscape (Koziol, 2012).

In the private sector, the mining industry in particular is at the forefront of restoration efforts. In large-scale restoration of complex habitats, a combination of direct seeding and plug planting is employed. For example, Alcoa's Jarrah forest restoration programme in Western Australia (Koch & Hobbs, 2007) and Rio Tinto's Littoral forest restoration programme in Madagascar (Vincelette *et al*., 2007) have established seed banks to support restoration activities.

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Chapter 10

Traditional ecological knowledge, traditional resource management and silviculture in ecocultural restoration of temperate forests

Dennis Martinez

Chair, Indigenous Peoples' Restoration Network (IPRN) of the Society for Ecological Restoration International (SERI), Member, Indigenous Peoples' Biocultural Climate Change Assessment Initiative (IPCCA) Steering Committee, Co-Director, Takelma Intertribal Project (TIP)

This chapter presents a broad-based overview of how traditional ecological knowledge (TEK) and traditional resource management (TRM) can inform ecological restoration and sustainable forest management, based on experiences from the temperate forests of far western North America.

We do not generally associate forest restoration with forest timber management; rather, we think of restoring degraded wild ecosystems to some semblance of their former healthy state. In this chapter it is argued that silviculture, a form of agriculture, can be enhanced by restoration and, reciprocally, appropriate silviculture can assist in restoration and maintenance of forest. The chapter concludes with the presentation of some insights into the genetic implications of silviculture, restoration and indigenous TRM and genetic relationships that can be affected either negatively or positively by how we manage both restoration and silviculture.

The relationships between restoration, silviculture and indigenous TRM are not well understood. While Western managers and ecologists frequently express interest in local examples of TEK, e.g. plant or animal indicators that could assist them in their research, it will be necessary here to take a more universal approach. This paper adopts this broad, holistic perspective in order to clarify these relationships and to bolster the argument for the importance of TEK and TRM to restoration and silviculture. To this end it is necessary to first describe indigenous TEK/TRM and the key, nearly universal, cultural practice of prescribed burning. The extent and ecological importance of indigenous burning is still controversial, but it is foundational to the argument for the use of an historical indigenous-managed forest model with which to guide restoration and enhance silviculture.

Indigenous cultural land-care practices or TRM, in concert with natural processes, created and maintained distinct cultural landscapes that could be described as a kind of indigenous agroecology or agriculture. These systems, which included modifying vegetation by fire, were employed over millennia to enhance ecosystems in order to produce food, medicine, cordage, basketry, cages and traps, ceremonial items, clothing, games, musical instruments, tools and utensils, weapons, fishing and hunting gear and structures (Anderson, 2005). The forest was (and still is for many indigenous peoples) the local supermarket, pharmacy and hardware store. Indigenous agroecology, like Western agriculture, influences the local availability, abundance, composition and distribution of plants (and, in the case of agroecology, animals). It is roughly equivalent to Western agriculture without the need for ploughing, fertilizing or irrigation, and without ecologically harmful side effects such as excessive nitrification and dependence on fossil-fuel inputs. It is not merely a kind of "proto-agriculture" representing a late phase in the evolution of what we conventionally understand as "true agriculture." It had proven its worth as the most adapted kind of management for the environments in which it evolved. It is questionable whether it would have had any need to evolve further.

TEK is a belief, knowledge and practice complex (Berkes, 2008) passed orally from generation to generation and informed by strong cultural memories and sensitivity to change. It encompasses a wide variety of ecological knowledge, including animal behaviour and social ecology, indicator species, weather prediction, fire behaviour and prescribed burning, gathering, fishing and hunting knowledge, relationships between insects, birds, plants and animals, agroforestry, agroecology, horticulture and memories of significant weather and other ecological events. Much of this knowledge is encoded in indigenous languages; when a language is lost, so is valuable ecological knowledge. Community knowledge specialists guide and regulate resource use, while families and clans exercise ownership management and conservation responsibilities for their particular places, thus avoiding the tragedy of the commons. Reciprocity, sharing and restraint are informed and maintained by a spiritual belief system with dire consequences (shame and misfortune) for those who are greedy and disrespectful toward the animals and plants that they regard as relatives in a kincentric world. Kinship is the glue that holds it all together. Traditional indigenous societies are conservative to their core and highly risk averse. To paraphrase what the International Indigenous Commission (IIC) told the delegates at the 1992 Rio Earth Summit, indigenous production methods involve increasing biodiversity by constantly creating new diverse habitats or niches – most often with intentional use of fire that maintained a fine-grained, patchy landscape mosaic – while maintaining surplus biodiversity or overcapacity as an untapped capital reserve. This brief summary of the context in which TEK is rooted provides a good segue to the main focus in this paper: TRM as just one of many possible components of TEK.

This chapter focuses on cultural land-care practices that contributed historically to a particular forest structure and composition (Society for Ecological Restoration International Science & Policy Working Group, 2004) and how this unique forest structure maintained by indigenous peoples can inform the spatial arrangement of subsistence and commercial timber and non-timber species. For example, modified historical indigenous models can be applied to even plantation forestry, modifying the spatial structure, making it more diverse, while sequestering carbon or providing timber and non-timber products. Special attention will be paid to forest genetics while integrating modified forest structure with native composition, i.e. how to reconnect commercial and/or subsistence forests with ecosystem function and resiliency in a time of rapid environmental change.

What we conventionally call "novel," "natural" or "pristine" landscapes are often, in part, degraded cultural or agro-ecological landscapes (some indigenous peoples call these landscapes their "garden"). Here is where the line between ecological restoration and restoration of cultural landscapes becomes blurred, requiring a different restoration term – "ecocultural" or "biocultural" restoration.

Ecocultural restoration is the process of recovering as much as possible of the key ecosystem structure, composition, processes and function that existed prior to European contact, along with traditional, time-tested, ecologically appropriate and sustainable indigenous cultural practices that helped shape ecosystems and cultural landscapes (Keenleyside *et al.*, 2012). This is done while simultaneously building in resilience to future rapid climate disruptions and other environmental changes (Box 10.1) in order to maintain ecological integrity in a way that ensures the survival of both indigenous ecosystems *and* cultures, including culturally preferred species – a distinguishing feature of ecocultural as opposed to ecological restoration (Martinez, 2013). It should, however, be noted that mostly non-cultural plant communities in which the cultural plants occur are also valued as relatives deserving of protection, restoration or both. Indigenous TEK and TRM and the resulting historical forest structure and composition managed by indigenous peoples can inform ecocultural restoration by supplying an initial reference model or baseline and can provide a way to bridge TEK/TRM and silviculture (Egan and Howell, 2005).

Ecological/ecocultural restoration is not, as is commonly believed, going back to some pre-industrial snapshot in time. Nor does it mean continuing with just the present degraded forest (also a snapshot in time). Rather than turning the clock back, we are resetting the evolutionary clock and attempting to restart a trajectory bounded by conceptually reconstructed historical ranges of variability in the types, intensities, extents and frequencies of natural disturbances or stressors, with which the forest ecosystem is genetically and ecologically familiar (Perry, 1994). When one considers the length of time indigenous peoples have been on the American continent (estimates have been consistently rising over the past century from a few thousand years to 30 000 years or more [Dobyns, 1966; Fiedel, 2000; Mann, 2005]), indigenous stewardship surely has affected forest genetics through selective harvesting and the use of fire that influenced cultural plant and animal abundance, characteristics and distribution (~300 plant species were typically utilized as well as many non-useful species affected by larger hunting fires).Therefore the term "natural" should include indigenous caregivers as a keystone biotic component of ecosystem dynamics. We hope, in ecocultural restoration, to at least be able to capture key features of disturbance regimes, structure, composition, processes and function together with longstanding cultural land-care practices and important cultural species (Box 10.1).

The reconstructed reference model is only a guide, but one that is anchored in real ecocultural and historical time. In the process of setting restoration goals, we will have to balance historical fidelity to the reference model with ecological functionality, resilience and integrity given changed environmental conditions (Higgs, 2003). But the model will assist in giving us a sense of direction by restoring an evolutionary trajectory that has been seriously derailed. This historical baseline is important for gauging environmental change and the degree of degradation. It is what we are striving to restore – even if we are not entirely successful or the work completed. Indeed, restoration will probably always require some periodic human intervention, such as controlled burning.

Contact with Europeans and their diseases killed up to 90 percent of the indigenous population in many places and created the common misconception of historically low populations. Indigenous populations were relatively large before contact with Europeans[17] and required prodigious amounts of material from plants that were burned the previous year, e.g. fire-induced epicormic and adventitious shrub or tree sprouts used in basketry, or the burning of sometimes hundreds to thousands of hectares to rejuvenate brush species (*Ceanothus* spp., oak, plum, hazel, mountain mahogany etc.) palatable to black-tailed deer (*Odocoileus hemionus columbianus*) and elk (*Cervus canadensis*) (Lewis, 1973; Boyd, 1999; Stewart, 2002; Blackburn and Anderson, 1993).[18]

17 Henry F. Dobyns, cited by Mann (2005), estimated the population of the Americas in 1491 at 90–112 million, compared with an earlier estimate by Mooney (1928) of 1.2 million for North America.

18 To give the reader an idea of the amount of burned plant material required, consider the following: in California, 35 000 stalks of milkweed (Asclepias sp.) or Indian hemp (Apocynum cannabinum) were required for one deer net about 15 m long (Blackburn and Anderson 1993) and 1200 sprouts of sourberry (Rhus trilobata) were needed for a burden basket. Twenty-five basket weavers in a typical California village of about 100 people might harvest about 250 000 shoots in a single season. Lightening could not be relied on to start the necessary fires because it strikes at random (i.e. it could not be relied on to strike where it was needed on a regular basis and was relatively rare in lower elevations and coastal areas) and because fire started by lightening was often different from fires started deliberately in terms of spatial selectivity, extent, frequency, intensity and seasonality. Sixty percent of cultural items came from plant material (Anderson 2005; Chester King in Blackburn and Anderson 1993).

Box 10.1.
Suitability of germplasm to site

Indigenous peoples are rarely in a position to assist migration of climate-vulnerable species because their territories are mostly relatively small and peoples are rooted in place, without the opportunity to follow displaced species except to higher elevations in some places. However, these places are relatively small in extent. Some cooler micro-sites in mountainous regions do occur at lower elevations. Individuals from thermally stressed species, such as the endangered keystone tree species, whitebark pine (*Pinus albicaulis*), have become well established on these sites (C. Millar., 2012, personal communication). Important cultural plants will have to be maintained on the reservation, rancheria or reserve, reinforcing the importance of historical reference models for indigenous peoples. Cultural "resistance" will be necessary to maintain cultural integrity through the adaptive suitability of the germplasm of cultural species. The indigenous oral tradition suggests that indigenous people have had to adapt to environmental change many times before. For example, oral tradition tells us that indigenous peoples moved salmon spawn in wet moss when rivers were blocked in Pacific Northwest North America (Sproat 1868; Campbell and Butler, 2010; Coast Salish Nuuchalnulth oral tradition). The last time this happened was in 1913 at Hells Gate, when a massive landslide blocked the Fraser River in British Columbia, Canada, and the Salish people built a flue around the slide to save sockeye salmon returning to spawn. Eventually, some of these cultural species may be displaced, but it is a question of buying time to allow alternative species to become available from adequate planning and not by default at the last hour.

Ecosystem-based adaptation is critically important; global warming and climate weirdness are already having an impact on indigenous peoples and the vulnerable ecosystems they inhabit. The challenge is to find sufficient genetic diversity in culturally important species. We will have to seek out adapted plants – isolated individuals, populations and subspecies – in addition to the cross-breeding currently being done. Forested landscapes with considerable heterogeneity may provide a number of possible micro-sites that could enhance forest refugial capacity. A good course of action would be to collect propagules from populations in extreme micro-sites (exposed to extremes of weather etc.) and propagate them in quantity in nurseries or cold frames for later transplanting back to the place of their origin or to similar micro-sites elsewhere. Assisted regeneration could also include comparisons of growth and survival of propagules from both extreme and non-extreme sites in standardized greenhouse conditions to analyse genotype differences or separate genotypic from phenotypic characteristics.. Other adaptive characteristics could also be explored, e.g. trees with earlier or later flowering times than other individuals of a population, drought tolerance or disease resistance, or healthy conifers with particularly thick rugose bark, exceptional sap flow, good sapwood-to-heartwood ratios, or deep root systems for wildfire and bark beetle resistance. This is an area for much-needed genetic research.

Cultural landscapes in far western North America were created and maintained by periodic burning by indigenous peoples. This kept forest succession in an arrested state, producing a fine-grained, patchy vegetation mosaic (Lewis, 1973; Anderson, 2005). Fire had many ecological and human benefits, including nutrient cycling, better access to hunted animals, less groundwater lost through evapotranspiration, pest control, stimulation of plant regrowth, improved wildlife habitat, increased seed germination and seedling survival and reduction of hazardous fuels. In the wetter regions of the coastal Pacific northwest of the United States and coastal western Canada, patch burning for berries, habitat or baskets (Turner, 2010), among other reasons, had less effect on forest succession. Decomposer arthropods and fungi cycled nutrients, while forest gaps were created primarily by windfall trees and snow or wind breakage. Stand-replacing wildfires occurred rarely and only during long droughts. This was not because the dominant tree species – Sitka spruce (*Picea sitchensis*), western red cedar (*Thuja plicata*), western hemlock (*Tsuga heterophylla*) and Douglas-fir (*Pseudotsuga menziesii*) – were fire-resistant (they are not) but because of the extremely moist environment.

Perhaps the best way to approach the extent and ecological significance of burning by indigenous peoples is to think of numerous small to medium-sized patch-burns occurring every one to 15 years or so and scattered across the landscape. The cumulative ecological effects of these frequent low to intermediate disturbances on ecosystem productivity and biodiversity were amplified by the frequency of these fire events.[19] These numerous, regularly burned patches exponentially multiplied ecotones, maintaining high biodiversity and quality wildlife habitat (Anderson, 2005). However, it was not necessary to burn everywhere. Indigenous peoples were very aware of the need for unburned wildlife cover and for protecting shade-adapted plants. Fuel-breaks were made and ridge tops were kept open to check fire spread into neighbouring watersheds, with backburns employed to protect vegetation and leave thermal cover for deer and elk. Burns in previous years acted as fuel-breaks for burns in the current year. Some sacred places also escaped burning, as well as selected brush-fields left to senescence before being harvested for fuelwood (Chester King in Blackburn and Anderson, 1993).

Burning was highly selective. It was typically performed in those vegetation types that produced significant amounts of cultural plants, were prime wildlife habitat and high in species richness: riparian zones, wet and dry prairies, wetlands and marshes, pine and oak savannas and woodlands and mountain meadows. Burning was also extensively utilized to create and maintain small to medium-sized gaps and larger meadows in relatively resource-poor forest types such as those dominated by coast redwood (*Sequoia sempervirens*) and Douglas-fir, e.g. the "Bald Hills" of coastal northern California (Lewis, 1973; Bonnicksen *et al.*, 1997), which have now lost over 30 percent of their former extent as a result of invasion of Douglas-fir since circa 1910.

What is the relevance of the fire-maintained forest to modern restoration and silviculture? We can begin to answer this question by considering the structure of the firescape managed by indigenous peoples. If one counts old-growth conifer stumps on west, south and east slope aspects in much of the interior, one frequently finds approximately 20 to 65 stumps per hectare in clumps, compared with 10 to 80 times that number of trees in a mid-successional state at present (Martinez, personal observation). With less competition from younger trees killed by repeated fires and with more sun, many old-growth conifers were practically almost open-grown, with full crowns extending close to the ground, structured like a carrot (called "grouse ladders" or "wolf trees" by loggers).

Fire set by indigenous peoples, and to a much lesser degree fire started by lightning strikes, was the main architect of forest structure and composition, favouring important fire-adapted species. One should think of the pre-industrial forest as a slowly changing assemblage of multi-aged species, including a mix of dominant mature and old-growth hardwoods and conifers, with fire recycling all seral stages of vegetation development at the landscape scale (Senos *et al.,* 2005) – a kind of relatively stable "steady-state shifting mosaic" (Perry, 1994). It is important not to conflate post-harvest early successional vegetation, mostly a diverse and unstable mix of (frequently introduced) annuals and short-lived perennials or shrubs, with more stable long-lived native perennial bunchgrasses, forbs and shrubs that are periodically renewed by intentional burning. Think of a slow turnover or shifting of early, mid- and

[19] In addition to fire, these disturbance events included a number of horticultural techniques, including pruning and coppicing, weeding, planting, seed sowing, tillage (women regularly dug in numerous tracts to harvest a variety of geophytic corms called "Indian potatoes") and erosion control (Anderson 2005; Turner 2005).

late-successional species together at one time and in one forested landscape.

What are we restoring? Our indigenous reference model suggests a variable forest structure with well-spaced conifer and hardwood trees and tree clumps mixed with patches and irregular colonnades and corridors of more-closely spaced trees and shrubs, with all age classes of most historical species represented (Anderson, 2005; Senos *et al.*, 2005). The newly created openings of varying sizes would be repopulated (either by natural regeneration if propagules remain on site or by replanting and/or reseeding/plugging; see "Ecological anchors" in Box 10.2) with restored bunchgrasses, forbs, and shrubs, approaching the historical species-rich understorey and meadow

Box 10.2.

Insights on diversifying a gene pool and restoring biodiversity through ecocultural restoration forestry and ecosystem-based adaptation to climate destabilization/ global warming in temperate far western North America

Methods and strategies based on traditional ecological knowledge (TEK) and traditional resource management (TRM)

Kipuka **strategy:** *Kipuka* is a native Hawaiian term used to describe a rock outcrop that lava spewing from volcanoes goes around instead of covering. Used in the context of ecocultural restoration, it suggests repeated islands or groups of trees, shrubs, ferns, forbs and grasses, i.e. the fine-grained landscape created and maintained through judicious use of fire. These patches and openings, including meadows, range from a size that is equivalent to the height of surrounding trees (patches) to as large as several hectares (meadows) depending on site conditions, elevation, forest type and restoration objectives. Objectives are not limited to trees. They include species-rich understories and meadows. While some – but by no means most – timber harvesters leave irregular islands to imitate fire effects, kipukas are more about the actual restoration of firescapes, not their imitation. Fire cannot be imitated in most of its effects on soils and vegetation. This is as much about restoring composition as structure. For example, spot-burns or pile-burns are often done in openings following thinning. These small patches will sometimes gradually fill with bunchgrasses and forbs seeded in the ashes. These kipukas, together with the hit-and-miss effects on understorey herbaceous vegetation of low-intensity burning, contribute to forest floor heterogeneity. These species-rich openings and meadows can be part of irregular herbaceous corridors that could be described as "flower trails" that guide pollinators and seed-carriers across the landscape (Anderson, 2005). Instead of one-time irregular structural manipulations by timber harvesters, kipukas are meant to be periodically burned. Frequent interventions have the cumulative effect of increasing species richness and diversity.

Variable density management or variable green retention: This is a method developed by forest ecologist Jerry Franklin. The objective is to "release" future old-growth and commercial trees from competition by smaller trees and brush, resulting in repeating sunny openings alternating with repeating areas of thick shady to partly shady vegetation. (The proportion of shade to sun extent will depend on forest type and site moisture regime.) In a relatively homogenous stand, the seedlings, saplings and poles with the fullest crowns and the largest diameters are retained, as are deformed trees, slow growing trees or trees on harsh sites. In the first case, it is hoped that these will become very large, healthy trees that will reproduce their superior characteristics over time. In the latter case, it is hoped that at least a few of the poorer trees will possess genes for exceptional drought and heat tolerance, disease resistance etc., and that these genes will be reproduced and perhaps multiplied in the forest over time. (For selection criteria for herbaceous understorey plants, see "Ecological anchors," below.)

Box 10.2. *(continued)*

Insights on diversifying a gene pool and restoring biodiversity through ecocultural restoration forestry and ecosystem-based adaptation to climate destabilization/ global warming in temperate far western North America

Redundancy: Just as good engineering requires structural redundancy to ensure safe structures in case a component fails, so environmental components – such as vegetation spatial combinations, closed–open and sun–shade contrasts, wildlife guilds, prey–predators, pollinators–seed carriers, flowering plant diversity, food webs, down wood and snags and compositional diversity – are repeated across the landscape. If one component fails, others of a similar class can take up the slack. Redundancy or risk-spreading is a principal goal of VDM.

Landscape heterogeneity: This concept is more than just genetic, structural and compositional diversity. Landscape heterogeneity means that managers actually look for and map unique micro-sites that are harsher and warmer and that could serve as possible sources for individuals, populations and subspecies that are better adapted to global warming. Random sampling of a particular species is not as likely to pick up adapted plants as doing a stratified and focused field meander that may reveal populations better adapted to harsh or hot sites. For example, United States Forest Service researcher Connie Millar, working in California's Sierra Nevada mountains, has noted a downward movement of some endangered white bark pines, finding seedlings established in cooler lower elevation sites. There are many of these micro-sites, especially in areas like the Coast, Cascade, Sierra Nevada or Klamath mountains that are topographically diverse. Restorationists will have many opportunities in projects to find unique heterogeneity in micro-sites, such as tree windfall, slash piles, large down wood, topographic depressions, mesic or very dry places, stream banks, rocky outcrops etc. (This is discussed in more depth in Box 10.3 in the context of building-in resilience to change.)

Ecological anchors: This is a method developed by Canadian forester Herb Hammond. An ecological anchor is any environmental component that will assist managers working in more homogeneous stands (e.g. tightly and uniformly spaced plantations or typical dense mid-successional forests) to determine which trees to thin, and those working in less dense forests with the need for increasing understorey genetic and compositional diversity. Examples include sun-loving herbaceous species that are culturally preferred, ecologically significant or endangered species that are being shaded out by trees, or culturally and ecologically valuable hardwoods still in the stand (e.g. oaks) that need release from overtopping conifers. Trees that are shading out these species would be thinned to allow the understorey to recover.

Conversely, trees protecting important shade-adapted species in the understorey would be retained. This method puts as great an emphasis on ecology as on merchantability in determining which trees to leave, i.e. removal of a conifer to release oaks or leaving a conifer as an anchor for shading even though it does not necessarily possess good or the best merchantable qualities.

Stepping-stone habitats for linking conservation reserves with forest matrices and providing connectivity: Conventional conservation wisdom divides forested landscapes into two spheres: reserves and the matrix surrounding the reserves that is sacrificed to timber. In fact, the matrix probably already has good-quality habitat that could be linked within the matrix and to nearby reserves, providing connectivity.

Reserves alone generally do not possess sufficient topographic and other kinds of diversity for wildlife habitat and for climate refugees. Linking up to the matrix could amplify good habitat and connectivity, provide cooler micro-sites that could increase

The refugial capacity of the forest to harbour plant and animal climate refugees and contain harsher or warmer sites that could provide adapted propagules for ecosystem-based climate adaptation. It may also facilitate gene flow between reserves and matrix, and between sources and refuges (Society for Ecological Restoration International Science and Policy Working Group, 2004).

flora and providing opportunities for future non-timber and cultural products (Martinez, 2008 [unpublished]). Forest ecologist Jerry Franklin calls this variable density management (VDM) or variable retention management (VRM) (Lindenmayer and Franklin, 2002). Depending on forest type, elevation, slope aspect and site conditions, a certain number of young and mature trees with good potential old-growth characteristics will be retained as future permanent old growth. This is in addition to trees with good potential for future commercial grade timber. Periodic management interventions (thinnings/harvests and prescribed fire) will be performed periodically over decades so that site conditions will not be changed too rapidly during any one intervention (see "Ecological Anchors" in Box 10.2).

Unlike standard silviculture with trees planted and/or thinned to regular and even grid-like spacing (especially in plantations), and with only one or two dominant, even-aged commercially valuable conifer species, it should be clear by now that the forest influenced by indigenous peoples is decidedly diverse, irregular and uneven-aged and fire-tolerant except for extreme weather-driven fire events (i.e. made up of numerous small even-aged stands from previous small fire events in an overall uneven-aged forested landscape; catastrophic fire events that we see today were extremely rare and always followed long periods of drought), with a species-rich understorey, the nature of which depends on whether trees are retained or removed. Cultural and other non-timber products mostly come from diverse forest understories (and from oak and pine woodlands, savannas, prairies and wetlands). This is a forest that is managed for both relative stability and productivity (both are a function of forest diversity) and for creating and maintaining a balance between forest use and conservation/restoration. The indigenous model (TEK/TRM) shows us that careful and ecologically informed use is a prerequisite for diversity and productivity. Indeed, forest use must further conservation and restoration as far as possible, while they in turn must sustain use and forest products. For more detail on how to balance silviculture/non-timber and cultural products with ecocultural restoration, see Martinez, 2008 [unpublished].

Instead of following United States and Canadian agency recommended bole-to-bole (trunk) tree spacing guidelines, crown-to-crown spacing (measured from outer foliage [crown] limits of one tree to outer foliage limits of another) provides for greater tree and tree grouping separation. As Canadian forester Herb Hammond notes, this makes for "better ecological choices for leaving trees than arbitrary stand density or basal area choices," including "habitat requirements for various [animal] species and maintenance of stand level diversity" (Hammond, 2009). The primary problem with relying on basal area (the total space occupied by tree boles per hectare) is that it is only a cumulative value and tells us little about their spatial arrangement in a particular place.

The sooner thinning occurs, the better chance trees have of achieving their genetic growth potential (usually by 30 to 40 years of age). Some remaining unthinned trees with sub-merchantable characteristics should be retained in case they have genes for exceptional drought and heat tolerance or disease resistance, or are good wildlife trees. While promising optimum characteristics of commercial and potential old-growth trees are important, indigenous peoples' holistic philosophy values the whole forest more than individual trees, i.e. not sacrificing biodiversity and wildlife habitat for optimum timber production. Cultural and commercial use must further conservation and restoration for the whole forest. This is the essence of TEK: reciprocity is required when using plant and animal "relatives." It is also important genetically: we may be sacrificing forest adaptive capacity to climate destabilization by eliminating too many non-commercial grade trees. For more detail on timber harvesting rotations, see Martinez, 2008 [unpublished].

Commercial harvesting is part of the VDM thinning process over several decades of multiple entries. Sustainable logging will continue but would be limited to harvest rotation cycles of 120 to 160 years for the entire stand, with parts harvested over shorter intervals within the stand. (Further north, harvest rotations for boreal forest should be considerably longer.) While timber volume may be reduced, longer rotations will ensure sustainability over the long term, while prescribed burning will reduce the cost of wildfire control and timber losses by significantly reducing hazardous build up of fuel. Sufficient numbers of trees must be retained to replace those lost through harvesting and natural mortality, using ratios ranging from 3:1 to 5:1, depending on forest type and site conditions (Martinez, 2008 [unpublished]).

Fire-hazard-reduction goals require the removal of ladder fuels, i.e. small and intermediate trees that can carry ground fires into canopies. Trees of different ages and sizes need to be segregated to break up contiguous fuels. The stand structure will, for the most part, be even-aged groupings in an overall uneven-aged forest. This is in fact the historical forest that resulted from frequent low to moderately severe fires. Each discreet tree grouping dated from a different small fire event. Succession arrested by indigenous practices in interior forests favoured earlier successional conifer species such as pines and mid-successional tree species such as Douglas-fir – valuable commercial species today. Advantages of earlier successional species for fire-hazard reduction include lower crown bulk densities (less biomass weight per cubic metre of foliage), self-pruning that removes fire-vulnerable lower branches and deeper feeder roots that can avoid excessive soil heating. Ground fuels were regularly consumed by burning by indigenous peoples, with charred large down wood sometimes lasting a very long time. Frequent low- to moderate-intensity fires leave a long-lasting (3000 to 12 000 years) legacy of charcoal that gradually mixes into the top metre of soil and sequesters carbon as well as providing cation-exchange sites, increasing forest productivity (Deluca and Aplet, 2008).

Regular fire fertilized the forest by cycling nutrients and when combined with reduced competition from smaller trees and brush, allowed the genetic potential for optimum growth to be realized. Managed burning contributed, along with lightning-ignitions, to the healthy old-growth giants that were used to build our cities. Prescribed fire directly assists silviculture and reduces or eliminates the need for broadleaf herbicides to control competing deciduous plants. A healthy ecosystem supports healthy timber, and ongoing sustainable timber harvesting and fire-based silviculture in their turn contribute to the maintenance of restoration by repeated harvest thinnings in perpetuity, with prescribed burning acting as the principal architect of forest structure. Reconnecting timber harvesting with ecosystems means, in part, reconnecting with indigenous fire regimes. Reconnecting with indigenous fire regimes means reconnecting with TEK and TRM, acknowledging the environmental legacy of indigenous peoples and its relevance today, and recognizing the environmental conditions that influenced the genetic structure of many species over a very long time as they co-evolved with indigenous fire practices and other disturbances – human and otherwise. As ethnobotanist Kat Anderson writes: “Landscapes are not just assemblages of species; rather, they are expressions of human evolution and species behavior. The adaptation of plants and animals that exist today are responses to past sequences of environmental conditions” (Anderson, 2005). Those past sequences were induced in large part by indigenous burning practices.

Local and traditional ecological knowledge based on qualitative observational approaches and Western experimental and quantitative approaches are increasingly being seen as complementary. As climate disruption continues to affect ecosystems and cultures at multiple spatial and temporal scales, observational data on sites that are not easily manipulated experimentally are becoming critically important. Even research sites that appear environmentally similar can be different enough to compromise experimental results. There is a real possibility of climate disruption exacerbating already degraded ecosystems, causing them to cross potentially irreversible thresholds or tipping points well before we are aware of it happening (Herrick *et al.*, 2010). It is for these reasons that place-based indigenous peoples are in a privileged position to maintain and monitor conservation and restoration in their homelands, ground-truth Western science's more generalized experimental and remote technological approaches and contribute to a more sustainable and biodiverse silviculture.[20]

[20] For example, in 1979, Western scientists using passive microwave technology discovered that Arctic sea-ice was losing extent and thinning, yet Inuit and Iñupiat peoples knew this in the early 1960s – approximately 15 years earlier.

Box 10.3.
Extent and nature of gene flow across fragmented agro-ecosystems

Gene flow is facilitated by variable density management, which is based on thinning that recreates the clumpy nature of forest trees of variable sizes under traditional resource management, including enough open spaces between tree groupings to allow "exchange of alleles among individuals and populations" (Friederici, 2003). This box focuses on the ponderosa pine forests of the southwestern United States, but the same principles of free gene flow also apply to many other overstocked forest types in far western North America – all influenced historically by indigenous burning regimes. While some geneticists maintain that conifers do not generally have a problem with gene flow, most forests in far western North America are impenetrably dense, with stocking rates as high as 7000 trees/ha or more. This is very likely to result in different patterns of gene flow compared with the more open and clumpy nature of the historical forest under indigenous management, and before effective fire-suppression policies began in the early twentieth century.

These small clumps are "genetic neighbourhoods". Dendroecologist Joy Nystrom Mast (in Friederici, 2003) explains:

"A group or clump of half-siblings is often created by a single older tree in the clump, but pollinated by different trees. As a result…unlimited pollen movement and hence gene flow among clumps helps prevent detrimental levels of inbreeding. Any highly inbred seedlings are subject to reduced reproductive rates, growth, and survivorship, and are usually outcompeted by outcrossed individuals… thereby reducing future levels of inbreeding within clumps."

Older trees became established and competed in environmental conditions different from their offspring – more-open stand conditions for older trees and more-crowded conditions for younger trees. This difference affects allele diversity so that, if conditions change and little gene flow occurs between older and younger trees, future adaptive capacity to changed conditions may be lost. Thinning too heavily can lead to loss of low-frequency or rare alleles. Thinning too lightly could prevent unimpeded gene flow between clumps. Clump spacing should be of a distance appropriate to forest type and density so that gene flow is neither too hindered nor too free. Younger trees should be maintained along with older trees. Indeed, forest restoration prescriptions should specify that representatives of all age-classes and all native species be retained on site (unless they are very invasive generalist native species that are already abundant). This is an area for genetic research. How much distance is required between clumps of which forest type, so that gene flow can occur while preventing loss of low-frequency or rare alleles? It should be noted that more than one entry will be necessary to approach pre-industrial forest structures. Multiple entries over decades are usually necessary in order to change forest environments at a rate that trees and other species can adapt to appropriately in the future. Therefore another question is: how much should be thinned in one entry in what forest type? Can genetic research help here?

Knowledge gaps and possible fruitful genetic research

Considering the importance of regular intentional burning, one wonders how both burning and selective harvesting of plants may have altered their genetic structure, much as plant breeders do today through selective cross-pollination. This was not proto-agriculture; rather it was indigenous agro-ecology that, like Western agriculture, influenced the local abundance, availability, composition, distribution and characteristics of plant (and animal) species. The cumulative effects of frequent burning of small patches carried fire effects to much of the forest. Frequent, low-intensity burning needs to be studied in order to reveal its effects on forest productivity and genotype selection of culturally favoured tree species.

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Chapter 11

Designing landscape mosaics involving plantations of native timber trees

David Lamb

Centre for Mined Land Rehabilitation, University of Queensland, Australia

Reforestation is often seen as a necessary part of any rehabilitation process once land becomes degraded. Depending on how it is done, reforestation can improve biodiversity conservation, stabilize hill slopes and improve watershed protection. However, designing any reforestation programme raises a variety of problems, particularly when several landholders are involved. This is because it is rarely possible to restore forest cover over the entire area, raising questions such as just how much reforestation should be done, what kind of reforestation should be carried out and where these new forests should be established. In most cases these questions are resolved through the actions of individual landholders acting independently and without reference to, or knowledge of, the planned actions of other landholders. Unfortunately, such an individualistic approach is unlikely to result in a satisfactory outcome. This is because many ecological processes, such as gene flow, operate at large landscape scales and the collective effect of many *ad hoc* decisions is unlikely to be as effective in restoring these ecological processes and functioning as a more strategic set of interventions that carefully target key localities and specify the type of reforestation carried out at each site. A more strategic intervention necessitates some degree of coordination across the landscape mosaic. This means that, in addition to how much, what type and where to reforest, a fourth question must be considered: how to organize reforestation on a landscape scale.

11.1. How much reforestation?

There is no simple answer to the question about how much reforestation is needed. It depends on how much natural forest remains and on the attributes of the biota that are present in the landscape and are vulnerable to extinction because of past deforestation. It will also be influenced by the land-use practices on the cleared land and on the socioeconomic circumstances of the people living in the area. Some private landholders may be interested in reforestation on part of their land but much will depend on the opportunity costs of doing so. A strong timber market or a market for ecosystem services (e.g. carbon sequestration) may increase the attractiveness of reforestation but only if the landholders believe they will benefit from it.

In the case of biodiversity conservation, numerous studies have shown that deforestation results in a loss of species proportional to the deforested area. However, once forest cover in the landscape falls below 20–30 percent, the spatial patterns and size of the forest fragments become more important in determining species survival than proportion of forest cover *per se* (Andren, 1994). Yet it is difficult to prescribe a minimum threshold target of forest cover for those undertaking reforestation. Different species have different habitat requirements; some will be affected by deforestation well before the forest area falls below 30 percent while others will persist even when the forest cover is lower. Perhaps the best that can be said is that more forest cover is better than less and that a landscape with a large area of forest will conserve more species and more diversity within species than one with less cover. It is usually difficult to predict how many species will return once a certain amount of restoration takes place. It is also usually not possible to specify whether a particular species will recolonize a particular site: much depends on the type of reforestation carried out and the quality of the habitats created.

11.2. What kind of reforestation?

The best type of reforestation for biodiversity conservation is that which is structurally complex and involves many native plant species. Some form of ecological restoration that eventually leads to the re-establishment of the former forest ecosystem (e.g. natural regeneration or multispecies plantings) would obviously be ideal. However, most industrial tree plantations use simple monocultures of exotic, fast-growing tree species because they generate a rapid financial return. Many smallholders also favour these simple monocultures when they grow trees for commercial purposes, although many farmers in the tropics also practise various forms of agroforestry that can involve a number of tree species.

However, it can be possible to have a much greater number of species present across a landscape even when the number of species at a particular site is small. This is because site conditions vary (necessitating the use of different species) and because different landholders have different goals or aspirations. Differences in site conditions and goals can lead to a mosaic of tree monocultures of different species and considerable landscape heterogeneity.

The wildlife species most likely to benefit from such monocultural plantings are those best described as habitat generalists. These are species able to utilize a wide variety of habitat types and they are rarely among those classified as endangered or vulnerable. A wider variety of species, including some with more specialized habitat requirements, can colonize monocultural plantations if these are grown on longer rotations and are not too distant from natural forests that act as sources of colonists. In these circumstances large numbers of tree species may eventually colonize the site (Keenan *et al.*, 1997). Initially these plants simply provide a structurally complex understorey but over time the colonists can grow up and add structural complexity to the canopy layers. This increases the value of the plantation as a wildlife habitat.

An alternative form of reforestation is to establish timber plantations containing several species. These may not have as many species as would be used in ecological restoration but, if carefully designed, can provide goods such as timber and non-timber forest products as well as habitats for some wildlife (Lamb, 2011). Their value in conserving biodiversity is further enhanced if any harvesting operations are infrequent. Such plantings are also likely to be more effective in stabilizing hill slopes and providing watershed protection than simple monocultures. Again, these may be colonized over time by further species if managed on long rotations and located near natural forest.

Different landholders are likely to have differing views on the merits of these various forms of reforestation (i.e. monocultures, multispecies plantations, ecological restoration) and, because of this, many landscapes could end up having representatives of all types of reforestation.

11.3. Where to undertake reforestation?

There are several ways of addressing the question of where reforestation efforts should be concentrated. Farmers interested in reforestation for commercial purposes may simply plant trees at sites not suitable for agricultural crops. Areas close to roads or timber markets may be especially attractive. Landholders more interested in reforestation for biodiversity conservation have two choices. One is to identify those areas where reforestation will help conserve small populations of species that are vulnerable to extinction. These might be isolated remnant patches of forest where the populations of some species are declining because their habitat areas are limited. Reforestation that enlarges these habitat areas could allow the populations of such vulnerable species to increase. A second approach is to increase the connectivity between remnant forest patches to allow the linkage of populations of species that are reproductively isolated from each other. This might be done by creating corridors between the patches of natural forest or by establishing small patches of forest within an agricultural landscape that might act as "stepping stones" and enable a species to move across that landscape between areas of natural forest. This would foster genetic interchange between the several populations and effectively increase the overall population size. As noted above, the type of reforestation undertaken at a site will influence which species can use the newly reforested areas. But even monocultures can be useful because they begin the process of creating a forest environment.

11.4. How to plan and implement restoration on a landscape scale?

All reforestation involves trade-offs and this is especially the case when it is being done at a landscape scale. Some landowners may be quite happy to reforest some parts of their land because the opportunity costs of doing so are low or because they are interested in the goods or ecosystems services that reforestation can provide. Others may be unwilling to undertake reforestation because they perceive the (opportunity) costs of doing so as being too high. Of course, individual farmers are not the only stakeholders involved. Other stakeholders include downstream water-users, wildlife conservationists, sawmillers and the broader community. Some of these are likely to have views that are quite different to those of local farmers, meaning it can be very difficult to get agreement on a reforestation programme that balances the wishes of individual landholders with those of the broader community.

Much land-use planning has been based on what might be referred to as a top-down approach. This often involves technical specialists working for a government agency and following certain prescriptions or guidelines. The advantage of this approach is that these planners can take a broad overview and make a judgment about what should be the best balance between competing interests in a particular area. Some sophisticated computer-based tools have been developed to assist these planners, including some that can be used to optimize conservation benefits (Chetkiewicz, St. Clair and Boyce, 2006; Millspaugh and Thompson, 2009; Thomson *et al.*, 2009). But this top-down, model-driven approach has a number of weaknesses. These include the fact that species differ in their habitat requirements and a reforestation programme that suits one species may be unsuitable for another. Likewise, the process must make arguable assumptions about trade-offs between different environmental benefits. Conservation of biodiversity is important but, for many stakeholders, so too is watershed protection or the maintenance of hydrological flows. Lastly, the process focuses on where to intervene but not on how to induce landholders to comply. It relies on compulsion (which is politically costly), compensation (which is financially expensive) or universal cooperation (which is improbable).

There is an alternative. Experience from many places suggests some kind of consultative planning process that incorporates both bottom-up and top-down approaches may be better than either approach alone. It may not lead to the most efficient design but it is likely to generate an outcome that is more acceptable to stakeholders and hence more likely to be maintained over time (Reitbergen-McCracken, Maginnis and Sarre, 2007). The main stages in such a process are as follows.

First, develop a landscape-level view of the problem. This involves gathering data about the existing biophysical and socioeconomic landscape mosaics, including the distribution of species, land ownership patterns, the economic circumstances of landholders and the trends in land use. Document the presence of rare, endangered or vulnerable species, together with information on threats to biodiversity conservation, such as invasive species or wildfires.

Second, engage with the stakeholders. Identify landholders and other stakeholders and obtain their views concerning future land-use practices.

Third, identify reforestation possibilities. Based on the preceding stages, develop a variety of reforestation scenarios that differ in the amount, type and location of reforestation activities. Evaluate the advantages and disadvantages of each scenario to the community and to individual stakeholders.

Fourth, decide on a reforestation plan. Consult with stakeholders and decide on a reforestation plan and timetable. This may involve using incentives or compensation to obtain the agreement of landholders occupying key locations (e.g. payment to landholders for the ecosystem services that reforestation on their land provides). It may also mean having to accept a suboptimal outcome for the sake of getting an agreement (an ideal restoration plan may have to progress in stages over a period of some years).

Finally, implement the plan, monitor the outcome and practise adaptive management. The final stage in any landscape restoration plan is to monitor it over time to ensure that the plan is actually implemented and that it generates the outcomes expected. Restoration can sometimes lead to unanticipated results and it may be necessary to intervene at a later date to ensure that biodiversity is indeed being conserved and that stakeholders remain supportive.

11.5. Will forest landscape restoration succeed in conserving all biodiversity?

From a conservation viewpoint, a landscape mosaic involving timber trees will have significant advantages over a homogeneous agricultural landscape. Even simple plantation monocultures surrounding small remnant patches of natural forest will help protect these from further disturbances and provide additional habitats for at least some of the species they contain. Corridors and small, scattered patches of trees, including monocultures, are likely to assist some species to move across an otherwise hostile environment. Landscape restoration also initiates a process of positive feedback in which wildlife, able to move across the landscape, assist in dispersing the seed of many plant species.

However, these types of landscape reforestation may not be enough to ensure the survival of all species. In the case of plants, those with large seeds are less likely to be able to be dispersed across landscapes either because they have no natural dispersal agent or because that agent is absent or present only in small numbers in degraded forests. The only way such species can be reintroduced to the landscape is to deliberately include them in revegetation programmes. The animal species of most concern are the habitat specialists, especially those occupying upper trophic levels and needing large home ranges. Partially forested agricultural mosaics are unlikely to be sufficient for such species and protected areas containing large areas of natural forest are likely to be the only way such species will be conserved.

11.6. Conclusion

It is difficult to develop reforestation designs that enhance the capacity of agricultural landscapes to conserve biodiversity. The problem is partly concerned with ecological issues but is largely to do with obtaining a consensus among stakeholders about the amount, type and location of any tree-planting. Apart from full ecological restoration, the best outcome would be extensive areas of multispecies plantations involving native species managed on long rotations. Such plantations are likely to provide a valuable complement to areas of natural forest that form part of a protected-area network.

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Insight 8

Identifying and agreeing on reforestation options among stakeholders in Doi Suthep-Pui National Park, northern Thailand

David Lamb

Centre for Mined Land Rehabilitation, University of Queensland, Australia

When the Doi Suthep-Pui National Park near Chiang Mai in northern Thailand was established in 1981 it contained a large population of Hmong people who had been living in the area for many years. These people initially practised shifting cultivation but, over time, had changed to more sedentary forms of agriculture. The villagers have neither Thai citizenship nor legal land tenure. Because of this they have had an acrimonious relationship with the park managers, who see them as illegal occupants destroying the conservation values of the park.

In order to resolve these differences and to plan a reforestation programme that would cover some of the deforested lands, members of the University of Chiang Mai organized and facilitated a meeting between park managers and the Hmong villagers (Elliott *et al.*, 2012). Both the villagers and park managers had full knowledge of the park and of the particular areas over which there was some disagreement. Where the opinions differed was in what should be done about the disagreements. On the first day, the facilitators met with National Park staff to determine their view of the problems and to seek ideas about a way forward. On the second day, the facilitators met with representatives of the village to seek their views. On the third day, the two groups were brought together. The Head of the National Park described what he saw as the problem and how the villager's livelihoods might be met in future. A representative of the villagers then gave their perspective on the problems they faced and on a way forward. Guided by the facilitators, discussions then took place on how these views could be reconciled. This included having the participants acknowledge (i) that forest conservation was something that both groups supported, (ii) that some cleared areas should be reforested to protect water supplies, and (iii) that villagers could continue to practise agriculture on some of the land currently being used but that their future economic opportunities lay with tourism and employment outside the Park.

Having achieved this common understanding, some prospective locations for reforestation within the Park were identified. This was done using maps derived from satellite imagery and GPS mapping prepared prior to the meeting. These showed the extent of the agricultural cropland, including orchard areas and annual cropping areas. Prospective reforestation areas were then identified on a laptop brought to the meeting. On the final day of the meeting a visit was made to the field where the alternative reforestation options were discussed. These discussions covered the extent of reforestation, the location of the areas to be reforested and the types of

Figure I8-1.
Location of the Doi Suthep-Pui National Park in northern Thailand

reforestation to be undertaken at each area. A final reforestation plan was then negotiated. This involved a programme of ecological restoration using native tree species based on techniques developed for the area over a number of years by Elliott *et al.* (2006).

Two factors in particular appeared to help make the process successful. One was that the facilitators were well known to both parties and had worked in the area for many years. Second, there were detailed maps showing exactly what each group had proposed. It was important that these could developed in time to be taken into the field on the last day of the meeting, where they gave participants confidence that they understood the trade-offs being made.

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Part 3

METHODS

Many restoration approaches and methods focusing on native species have been developed and fine-tuned over the years, reflecting the diversity of species and ecosystems, degradation factors, stages and socioeconomic contexts. In Part 3, some of the scientists who have developed these approaches or have been most active in promoting them describe some of the most widely applied and studied methods and their principles. In many cases these descriptions are complemented by case studies. The general methods are divided into those focusing on ecological restoration (Chapter 12) and those that also include production objectives for timber or non-timber products (Chapter 13), although the distinction between these two categories is not always clear and many of the methods yield systems that produce multiple benefits. Approaches used for restoring specific habitats and degradation conditions, such as mangroves, dry lands and previous mine sites, are presented separately, as these usually require specific attention on restoring not only vegetation but also soil properties and hydrology (Chapter 14). Finally, three approaches for restoring genetic diversity of particular threatened tree species are described (Chapter 15).

Figure 3.0.
Geographical overview of the main sites applying the methods presented in the study

Chapter 12

Ecological restoration approaches

12.1. Miyawaki method

Akira Miyawaki

Japanese Center for International Studies in Ecology, Institute for Global Environmental Strategies, Japan

The Miyawaki method (Miyawaki, 1993, 2004) was developed by integrating two concepts, the first based on the study of potential natural vegetation and the second derived from observation of Japanese sacred forests (*Chinju-no-mori*) renewed for centuries by monks, who planted seedlings of many species simultaneously.

The approach consists of planting seedlings of the maximum possible number of tree species that characterize the potential natural vegetation, from pioneer species to late-successional ones. From the day they are planted the seedlings change the ecology of the site, and the species and individual trees undergo natural selection through competition, resulting in the creation of a diversified natural forest.

The restoration process can be divided in four phases (Figure 12.1):

1. **Definition of the potential natural vegetation:** The potential natural vegetation is described by studying relict vegetation. Field survey data are subjected to descriptive phytosociological analyses, leading to the identification and mapping of potential vegetation units.
2. **Intervention planning:** This phase identifies the species required and determines the amount of planting stock needed to establish the forest. Surrounding areas are identified where the propagation material for the production of planting stock can be found.
3. **Execution plan:** This is divided into two stages:
   a) **Preparation** of the material and the site. The area to be restored is prepared by adding topsoil from surrounding native forests, straw and, where possible, components of the understorey vegetation of the neighbouring woods. Before planting, the planting stock is acclimatized for one to four weeks in the surrounding areas, either under the shelter of existing vegetation or under an artificial sheltering system.
   b) **Plant** seedlings that have extensive root systems randomly at high density (3–5 individuals per square metre) and mulch with straw or other organic materials.
4. **After-planting operations:** Weed and irrigate once or twice, if necessary, during the first two years.

The result is a diversified forest that is left to grow naturally after the first two years.

The most innovative element of the Miyawaki method is the application of the concept of "contemporary succession." This assumes that the

Figure 12.1.
The four phases of the Miyawaki method

Source: Miyawaki (1999).

native species normally associated with different successional stages, when planted simultaneously, generate an "assisted succession" (human-supported succession concept) that allows the development in a few decades of the relatively stable late-successional stage.

Planted simultaneously, all the species become part of a rapid succession. After the first phase of rapid growth, there is a natural selection of species (and individuals) best suited to micro-sites, and the plantation will evolve into a late-successional stage without the need for further action, through what can be described as a new succession theory (Figure 12.2).

Climate, soil and topography interact to create a certain type of climax forest. Human

Figure 12.2.
Comparison between Miyawaki's new succession theory and classical succession theory

Source: Miyawaki (1999).

intervention alters the plant cover. At this point two alternatives can be envisaged:

1. **Classical succession:** let nature take its course; wait two to ten years for the annual herb community to be replaced by perennial grasses, another 10 to 15 years for a community of shrubs to develop, 15 to 50 years for the heliophilous tree species to develop and finally 200 to 300 years or more for the late-successional species to establish.
2. **New succession:** simultaneous planting of seedlings (2–5/ m^2) of species belonging to the potential natural vegetation. This can give rise to a semi-natural environment very similar to a young late-successional forest within 40–50 years.

This is explained by positive interactions among the diverse species planted. Immediately after planting the microclimate changes, becoming more favourable for the young plants. Rather

Box 12.1. The Green Tide Embankment

"On March 11, 2011, Eastern Japan suffered major damage from the Great East Japan Earthquake and tsunami that followed. We conducted surveys on the disaster areas. The surveys proved that monoculture forests of fast-growing intolerant exotic tree species such as *Pinus thunbergii* (black pine) and *Pinus densiflora* (red pine) were almost destroyed and some were carried landward and extended damage by colliding with people, houses and cars. But forests of main and companion trees from the local potential natural vegetation stood firmly and exerted an influence on reducing the power of the tsunami. Main tree species of the forests are *Persia thunbergii* and evergreen *Quercus* (oaks), and companion tree species are also evergreen broadleaved trees including *Camellia japonica*, *Neolitsea sericea* and *Euonymus japonicus*.

"After the earthquake and tsunami, huge heaps of debris remained dispersed in the disaster areas. Debris is not industrial waste, but natural resources from the earth. After removing poisonous and inorganic objects, it should be used effectively. From our results in reforestation in the Brazilian Amazon, I suggest to the central and local governments, corporations and non-profit organizations that we should build mounds on the coastline of the disaster areas by mixing soil and debris, and plant indigenous tree species on them to form quasi-natural forests. Roots of plants also breathe under the ground. Mounds built of soil and debris have hollows and contain much air. Therefore trees can grow well. The forests on the mounds will function as a breakwater and protect lives and properties of local people from future tsunamis. I would like to build the Green Tide Embankment, 300 km long from the north to the south.

"The native forest system will last for 9000 years until the next glacial age, though there is alternation of individuals.

"Mature trees, which have grown large enough, can be cut selectively and utilized for furniture, architectural materials and other purposes. Forests coexist with local economies. After selective cutting, a successor replaces the harvested tree and the forest ecosystem will be maintained.

"Everywhere in the world, forests consisting of indigenous trees save lives and property of local people. Ecological reforestation based on the potential natural vegetation is indispensable in our safe living environments and regional economy. Let's extend the reforestation movement by planting indigenous trees proactively, from tropical rainforest regions to other areas of the world."

Description of the "The Green Tide Embankment" project from a keynote speech given by A. Miyawaki at the International Symposium on Rehabilitation of Tropical Rainforest Ecosystems at the Universiti Putra Malaysia, Malaysia in October 2011.

than suffering from competition from neighbouring plants, during the first months after planting the seedlings benefit from the positive effects (e.g. lower soil temperature during the day, windbreak effect or mitigation of extreme heat). Beneficial micro-site effects improve water availability and soil stabilization. Over time, natural selection will lead to the survival of the best-adapted individuals.

The Miyawaki method has been used in over 1700 sites around the world, on extensive areas as well as to establish windbreaks along roads and railways (Miyawaki, 1998). Since 1971, over 40 million native trees have been planted using this method. In 2012 Dr Miyawaki launched the Green Tide Embankment project, which is using the Miyawaki method to establish a green embankment all along the Japanese coast damaged by the 2011 tsunami to protect it against future events.

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12.1.1. Tropical rainforest rehabilitation project in Malaysia using the Miyawaki Method

Nik Muhamad Majid

Institute of Tropical Forestry and Forest Products (INTROP), Universiti Putra Malaysia, Malaysia

The Joint Research Project for the Rehabilitation of Tropical Rainforest Ecosystems was launched by Mitsubishi Corporation in 1991, with support from Universiti Putra Malaysia (UPM) and Yokohama National University (YNU), Japan. The project adopted the Miyawaki method (see above).

The goals of the project include developing techniques for the rehabilitation of degraded areas and conducting research to assess the health of rehabilitated forests.

Site information

The project comprises two sites, one in Sarawak and the other in Selangor, Malaysia.

The project was initiated in July 1991 on a 47.5 ha site on the Universiti Putra Malaysia Bintulu Campus, Sarawak (113°03′41.67″E; 3°12′32.28″N). The site previously had been badly degraded by shifting cultivation activities. The rehabilitation project in Bintulu was executed in four phases, and currently the site has several different-aged forests, the oldest being over 20 years old. These forests give researchers the opportunity to study various ecological parameters at different stages of forest growth.

In 2008, following the success of the Bintulu project, a new agreement was signed between UPM and Mitsubishi Corporation to establish a new model forest by planting indigenous tree species in an urban setting, using 27 ha of degraded land located inside the UPM Serdang Campus (101°43′32.27″E; 2°59′45.16″N). The establishment of this model planted tropical forest was initiated at a tree-planting ceremony on 26 November 2008 at UPM's Arboretum, which lies between the Kuala Lumpur–Seremban Highway, the Kuala Lumpur–Putrajaya Highway and the railway between Kuala Lumpur International Airport and the city. Formerly pastureland, this area was degraded by the construction of the railway track and six-lane highways.

Restoration activities

Malaysia is considered to be one of the world's leading mega-diverse biodiversity hotspots, with tropical rainforest covering an area of more than 7.6 million hectares, or about 70 percent of the country's total land area. The country is richly endowed with diverse flora and fauna that have the potential to be developed and utilized in various natural products and services. Forests are still the main source of income for the country. However,

TABLE12.1.
Number of species per family planted at the Bintulu and Serdang restoration sites, Malaysia

Family	No. of species		Family	No. of species	
	Bintulu	Serdang		Bintulu	Serdang
Anarcadiaceae	7	5	Lecythidaceae	1	2
Annonaceae	1	1	Leguminosae	4	6
Apocynaceae	4	1	Melastomataceae	1	–
Araucariaceae	–	1	Meliaceae	1	2
Bombacaceae	3	2	Moraceae	5	5
Burseraceae	3	2	Myristicaceae	1	3
Celastraceae	–	2	Myrtaceae	9	5
Combretaceae	–	1	Olacaceae	1	1
Compositae	1		Oxalidaceae	1	1
Dipterocarpaceae	57		Rubiaceae	–	2
Ebenaceae	2		Sapindaceae	4	6
Euphobiaceae	4		Sapotaceae	4	4
Fabaceae	–		Simaroubaceae	1	–
Guttiferae	5		Sterculiaceae	–	3
Icacinaceae	1		Thymelaeaceae	1	2
Irvingiaceae	–		Tiliaceae	–	2
Lauraceae	3		TOTAL	126	117

harvesting activities have caused serious degradation to the forest ecosystems.

A comprehensive research approach was initiated at the two sites to determine the extent of damage as well as the effectiveness of the reforestation and rehabilitation programmes. The area selected for planting on the Bintulu site was a coastal forest that included heath and lowland dipterocarp forests. Tree-canopy species were selected from the natural vegetation of a similar area to ensure the suitability of the tree species to the environment, based on assumption that indigenous species are well adapted to local conditions. Seeds and wildings were collected in Similajau National Park, Likau Forest Reserve, and the Experimental Forest and Arboretum at the UPM Bintulu Campus. Planting techniques were mound planting, open-area planting and partial-shade planting.

As of 2011, roughly 350 000 seedlings from 126 tree species have been planted in four different areas (Table 12.1). The species planted can be classified into three groups: light-demanding, shade-tolerant and slow-growing species. The light-demanding species include *Shorea ovata, S. mecistopteryx, Artocarpus integer, Pentaspodon motleyi* and *Whiteodendron moultianum.* The shade-tolerant species include *Shorea macrophylla, S. gibbosa, S. materialis, Hopea beccaariana, Cotylelobium burckii, Calophyllum ferrugenium, Parashorea parvifolia* and *Durio caranatus.* The slow-growing species include *Diospyros* sp., *Hopea kerangasensis, Palaquium gutta* and *Vatica* sp. In addition, over 100 research plots have been established in the rehabilitated area and the growth of the planted seedlings is regularly monitored.

A total of 19 500 seedlings, including rare and endemic species, have been planted on the

UPM Serdang Arboretum site using open-area planting. Tree species selection (Table 12.1) was based on vegetation studies in several forests in Peninsular Malaysia:

- Ayer Hitam Forest Reserve, a high conservation value forest of *Dipterocarpus crinitus* and *Hopea nervosa*, representing the lowland dipterocarp forest of Selangor.
- Semangkok Forest Reserve, representing lowland dipterocarp forest with *Shorea leprosula* vegetation association.
- Pasoh Forest Reserve, representing the lowland dipterocarp forest of Negeri Sembilan.
- Leban Condong Forest Reserve, representing the heath forest of Pahang.
- Rompin forest, representing the swamp forest of Pahang.
- Segari Melintang Forest Reserve, representing the *Shorea lumutensis* vegetation association since this species is an endemic in this forest reserve.
- Mersing forest, representing the *Shorea peltata* vegetation association since this species is endemic in this type of forest.

Project outputs

Project outputs to date include the following:

- Forest restoration: A mixed, virgin tropical rain forest has been recreated through human innovation (Figures12.3 and 12.4). The rehabilitated forest has attracted wildlife and many other plant species, and has improved soil fertility, the hydrological cycle and the microclimatic environment.
- Research: Many scientific papers have been published or presented at national and international conferences. This project contributed to UPM being ranked sixth among 95 universities in the world in the Green Metric World University Ranking 2010 for promoting sustainability through environmental conservation and green technology.
- Public awareness: Over the past two decades, at least 10 000 people have participated in planting ceremonies at the Bintulu project site, and another 2000 people have been involved in the Serdang project over the past four years (Figure 12.5). The events were widely covered by both local and international media, including the National Geographic Channel.
- Human capital development: The project has been the subject of six Ph.D. dissertations, seven M.Sc. theses and more than 20 B.Sc. theses.
- Linkages: The Acid Deposition Monitoring Network in East Asia (EANET), based in Niigata, Japan, has started a research project at the Bintulu site as one of its monitoring stations in the Asia–Pacific region to evaluate the effects of air pollution on forest ecosystems.
- Two international symposia were organized in 1991 and 2011 to discuss recent research findings and current issues related to forest rehabilitation and promote international collaboration among scientists, academics, policy-makers and forest industry stakeholders.

Figure 12.3.
Bintulu site before planting (1991)

Figure 12.4.
Bintulu site (2014)

Figure 12.5.
Planting ceremony at Serdang Site (2014)

12.1.2. Adapting the Miyawaki method in Mediterranean forest reforestation practices

Bartolomeo Schirone and Federico Vessella

AgroForestry, Nature and Energy Department (DAFNE), University of Tuscia, Viterbo, Italy

Natural forests in north Sardinia, Italy, have been degraded over centuries by human activities, such as livestock husbandry and wood exploitation. Since 1905, periodic attempts have been made to reforest the region using traditional techniques, mainly planting maritime pine (*Pinus pinaster* Aiton.), Aleppo pine (*Pinus halepensis* Mill.), Atlas cedar (*Cedrus atlantica* (Endl.) Carrière), cork oak (*Quercus suber* L.), downy oak (*Quercus pubescens* Willd.) and sweet chestnut (*Castanea sativa* Mill.). The trees were planted at low densities (300–2200 plants/ha) along contour lines after forming terraces by subsoiling, or across the slope in pits. Low planting density has traditionally been considered appropriate in arid and semi-arid environments to avoid competition for water resources between plants. However, there is little evidence that competitive processes outweigh co-operative processes, such as mutual shading, that can enhance seedling survival.

Experimental design

In May 1997, two experimental forest restoration plots were planted in Pattada (Province of Sassari, North Sardinia) to test the effectiveness of the Miyawaki method for reforestation. The

Miyawaki method involves planting both pioneer and late-successional species to a target density, often up to 10 000 or more seedlings/ha, and has been successful in reducing the time taken to achieve complete environmental restoration (see section 12.1). This is the first time that the Miyawaki method has been tested in Mediterranean Europe.

The trial was conducted by the University of Tuscia with logistical and monitoring support from the Regional Forest Directorate of Sardinia and political support from the Municipality of Pattada.

Both experimental plots were degraded and abandoned sites on which several reforestation projects had failed. A survey of the natural plant communities in neighbouring areas was conducted and the climate was characterized to evaluate the possible natural vegetation for the study sites and to select appropriate species for the reforestation. Seeds were collected from nearby natural forest stands and germinated in four greenhouses owned by the Regional Forest Directorate of Sardinia. Seedlings were grown in plastic bags for one year before being planted out in the field.

Slight modifications were made to the Miyawaki method. For instance, no new topsoil was added to the restoration sites, but soil was tilled to improve soil water storage over the winter and reduce water stress during the summer. Several mulching materials were used (sawmill residues, dry and green materials), and no weeding was done after planting. Local climatic conditions were analysed using climate diagrams developed by Walter and Lieth (1967) to determine optimal planting time.

Site A (40°37′32″N, 09°11′08″E, 760 m above sea level) covered 4500 m^2. Plot preparation consisted of clearing and tilling a series of 3.5-m-wide strips. Pot-grown tree seedlings were then planted at a density of approximately 8600 plants/ha. Site B (40°36′54″N, 09°10′04″E, 882 m above sea level) covered an area of 1000 m^2. In contrast to site A the entire plot was cleared and tilled. Planting density was approximately 21 000 seedlings/ha.

The plots were planted with both early-successional species and late-successional species to improve resilience of the plant community. On Site A, 1723 seedlings were planted, belonging to 22 indigenous tree and shrub species: 25 percent pioneers (e.g. *Arbutus unedo, Pinus pinaster, Spartium junceum* and *Myrtus communis*); 10 percent mid-successionals (e.g. *Celtis australis, Ligustrum vulgare* and *Pyrus communis*); and 65 percent late-successionals (e.g. *Quercus suber, Quercus ilex, Acer monspessulanum, Taxus baccata* and *Malus domestica*). On Site B, 2139 seedlings belonging to 23 autochthonous species were planted: 17 percent pioneers (*Arbutus unedo, Juniperus oxicedrus, Pinus pinaster* and *Myrtus communis*); 14 percent mid-successionals (*Celtis australis, Fraxinus orsnus, Phyllirea latifolia* and *Thymus vulgaris*); and 69 percent late-successionals (*Quercus suber, Q. ilex, Ilex aquifolium* and *Taxus baccata*).

Results

The plots were surveyed in 1998, 1999 and 2009.

By 2009 (i.e. 12 years after planting) early-successional tree species were well established, with stable populations, and the plots had a high level of plant biodiversity. Mean mortality rates for all species were 61 percent in Site A (672 plants survived) and 84 percent in Site B (336 plants survived; Table 12.2). The difference in mortality rate between the sites was mainly the result of poor drainage in Site B. The forest species that are most prevalent in local natural forest (i.e. maritime pine and the oak group) survived well in both sites, thus maintaining the possibility of achieving intermediate and late-successional vegetation stages. In addition, several indigenous species that had not been planted were found on the sites (e.g. *Erica arborea* and *Prunus spinosa*). The survey results suggest that cooperative processes (e.g. mutual shading) facilitated the establishment of some species, in particular the mid- to late-successional ones. The high planting densities adopted in the sites reduced, for instance, the impact of acorn predators, thus encouraging oak regeneration (i.e. the main late-successional forest species in Mediterranean environments) and favoured root anastomosis processes (connection of normally separated roots), which seems to influence the stability of the ecosystem and reforestation success (Kramer and Kozlowski, 1979).

While the experiment consisted of only two small field trials, comparison of the results, in

TABLE 12.2.
Survival of planted seedlings in two plots restored using the Miyawaki method. The seedlings were planted in 1997 (year 1) and evaluated in 2009 (year 13). Dashes indicate the species was not planted.

Species	Site A			Site B		
	No. of seedlings		Survival (%)	No. of seedlings		Survival (%)
	Year 1	Year 13		Year 1	Year 13	
Pioneer species						
Arbutus unedo L.	50	41	82	11	0	0
Juniperus oxicedrus L.	–	–	–	45	30	66.6
Myrtus communis L.	19	1	5.3	95	4	4.2
Pinus pinaster Aiton.	273	208	76.2	155	80	51.6
Rosmarinus officinalis L.	23	15	65.2	23	0	0
Salvia officinalis L.	5	0	0	4	0	0
Spartium junceum L.	74	29	39.2	21	0	0
Middle-successional						
Celtis australis L.	22	3	13.6	37	0	0
Fraxinus ornus L.	8	1	12.5	9	0	0
Ligustrum vulgare L.	126	29	23.0	13	4	30.8
Phyllirea angustifolia L.	1	1	100.0	–	–	–
Phyllirea latifolia L.	–	–	–	203	0	0
Pyrus communis L.	19	10	52.6	22	10	45.4
Thymus vulgaris L.	–	–	–	24	0	0
Late-successional						
Acer monspessulanum L.	21	2	9.5	30	0	0
Castanea sativa Mill.	42	1	2.4	–	–	–
Ilex aquifolium L.	112	23	20.5	125	0	0
Laurus nobilis L.	22	3	13.6	19	0	0
Malus domestica Borkh.	21	7	33.3	19	0	00
Quercus ilex L.	300	159	53.0	394	96	24.4
Quercus pubescens Willd.	268	116	43.3	93	8	8.6
Quercus suber L.	11	7	63.6	621	96	15.4
Sorbus torminalis (L.) Crantz	18	4	22.2	24	8	33.3
Taxus baccata L.	251	9	3.6	126	0	0
Viburnum tinus L.	58	3	5.2	26	0	0

terms of species densities and choices, plant biodiversity and ecosystem composition, with those of other reforestation practices traditionally applied in the same ecological context indicates some interesting differences in the growth performance of the species under the Miyawaki method. Traditional reforestation methods resulted in simpler vegetation structures (Table 12.3). Moreover, when traditional methods are used, growth performance of secondary species (measured by plant density and mean height) is severely reduced by the highly competitive shrub species (*Erica arborea* and *Arbutus unedo*) that occur spontaneously and in large numbers. In contrast, trees on the Miyawaki plots developed more rapidly. This was particularly the case for early-successional species.

TABLE 12.3.
Height of 12-year-old trees in four plots reforested using different methods. Sites A and B were established using the Miyawaki method, Site C was reforested using traditional pit planting (785 plants/ha) and Site D employed contour planting on terraces (1048 plants/ha). Dashes indicate species not planted, and zeros indicate planted species that did not survive in 2009. Data are given for species for which at least two individuals survived on each plot.

Species	Height (cm)			
	Site A	Site B	Site C	Site D
Pioneer species				
Arbutus unedo L.	32.7 ± 4.1	0	500.0 ± 35.8	110 ± 20.6
Cedrus atlantica Endl.	–	–	–	162 ± 54.6
Erica arborea L.	–	–	115.0 ± 12.7	130 ± 18.6
Juniperus oxicedrus L.	–	36.2 ± 18.5	–	–
Myrtus communis L.	10.0	10.0 ± 1.4	–	–
Pinus pinaster Aiton.	433.2 ± 143.6	325.5 ± 38.6	376.4 ± 73.0	425.7 ± 25.1
Rosmarinus officinalis L.	89.3 ± 33.9	0	–	80.0 ± 14.9
Spartium junceum L.	110.7 ± 62.2	0	–	–
Mid-successional				
Celtis australis L.	26.7 ± 28.9	–	–	–
Ligustrum vulgare L.	32.8 ± 52.6	30 ± 8.16	–	–
Pyrus communis L.	71.0 ± 65.1	60 ± 61.2	–	–
Sorbus torminalis (L.) Crantz	35.0 ± 50.0	40 ± 12.9	–	–
Late-successional				
Acer monspessulanum L.	40.0 ± 14.1	0	–	–
Ilex aquifolium L.	45.2 ± 30.6	0	–	–
Laurus nobilis L.	30.0 ± 17.3	0	–	–
Malus domestica Borkh.	100.0 ± 45.5	0	–	–
Quercus ilex L.	34.2 ± 32.1	40.8 ± 36.2	69.4 ± 23.2	146.2 ± 38.1
Quercus pubescens Willd.	23.6 ± 27.5	10 ± 5.3	–	–
Quercus suber L.	174.3 ± 49.6	77.5 ± 51.9	–	–
Taxus baccata L.	33.3 ± 38.0	0	–	–

Conclusions

Overall, this example of restoration using the Miyawaki method can be considered quite successful, even if some further improvements are required. For instance, early-successional species may have been planted in excessive numbers, thus competing with the intermediate- and late-successional species. Optimal planting density will have to be tested. An economic analysis should be performed to compare the costs of reforestation, including post-planting silvicultural practices, between traditional reforestation methods and the Miyawaki method. Planting costs when using the Miyawaki method are relatively high because of the high planting density and the associated labour requirements (even with non-specialized labour). On the other hand, the Miyawaki method requires no post-planting care such as weeding or thinning.

Even if costs of the Miyawaki method were are higher than those of the traditional reforestation techniques, the quality of forest achieved in a relative short time (i.e. after 12 years), would make it worth considering for use in protected areas and natural parks where traditional plantings are not easily accepted because of their aesthetic and ecological impacts. In traditional plantings, trees are placed in regular and fixed schemes, creating an easily recognizable artificial landscape, especially when using exotic species. The Miyawaki method, on the other hand, restores forest that is better integrated in the surrounding landscape because of its use of local species and a randomized planting scheme that evolves mainly according to the ecological and competitive processes among the species.

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12.2. Framework species method

Riina Jalonen[1] and Stephen Elliott[2]

[1] Bioversity International, Malaysia
[2] Forest Restoration Research Unit, Chiang Mai University, Thailand

The framework species method can be a particularly effective approach for restoring forest ecosystems where fragments of intact forest remain within about 10 km of restoration sites. In this method, selected indigenous tree species are planted on the restoration site to promote natural recruitment and succession. The planted trees shade out weeds to "recapture" the site and re-establish forest structure. They also reinstate ecological processes such as litter accumulation and nutrient cycling. These so-called "framework species" are selected for their ability to provide resources (e.g. nectar, fruit and nesting sites) at an early age. These resources attract seed-dispersing animals and thus facilitate dispersal of seeds of non-planted tree species (i.e. recruit species) into the site from forest remaining in the surrounding landscape. Improved site conditions (i.e. weed-free, humus-rich forest floor) favour germination of naturally dispersed seed and establishment of tree seedlings (FORRU, 2006; Figure 12.6).

Typically, 20–30 tree species are planted on each restoration site. Good framework species grow fast at seedling stage, rapidly develop large and dense crowns that shade out weeds, bear fruit at young age to attract seed-dispersing animals, and survive well in field conditions, including after fire where relevant (FORRU, 2008).

Figure 12.6.
Process of site restoration using the framework species method. Note the positive feedback loops that reinforce and facilitate ecosystem recovery.

Source: Redrawn from FORRU (2006).

Framework species should preferably include both early- and late-successional forest tree species to accelerate natural succession and facilitate the recovery of a complex forest structure. Many late-successional tree species can be planted since they perform well in the open, sunny conditions of deforested areas. Under normal circumstances they fail to colonize such areas because of lack of seed dispersal (FORRU, 2006).

Seeds or wildings of framework trees are usually collected from nearby forest. Ideally, they should be collected from as many parent trees as possible to ensure that a wide range of genetic diversity is captured. However, because the purpose of the framework species is to quickly recapture the site, phenotypically superior parent trees and propagated seedlings should be selected (Blakesley, Hardwick and Elliott, 2002). After propagation in a nursery, seedlings are planted on the restoration site at a typical average spacing of 1.8 × 1.8 m (approximately 3100 trees/ha). The seedlings must be tended for at least two years after planting by regular weeding. Other management practices, including fertilizer application, protection from wildlife and care of naturally recruiting seedlings, are also recommended (FORRU, 2006).

Studies have demonstrated the effectiveness of the framework species method. In a trial plot in Doi Suthep-Pui National Park, northern Thailand, 73 non-planted tree species had established eight to nine years after planting framework tree species (Sinhaseni, 2008). With 57 planted framework tree species, the total tree species on the site amounted to 130, equivalent to 85 percent of the total tree flora expected in an intact forest in similar area under the same conditions. Most of the tree species recorded had germinated from seeds dispersed from nearby forest by birds (particularly bulbuls), fruit bats and civets. The species richness of the bird community also increased from about 30 species before planting to 88 after six years, representing about 54 percent of bird species recorded using the same methods in nearby intact forest (Toktang, 2005). The species richness of mycorrhizal fungi and lichens has also been reported to increase dramatically in the restored plots, often exceeding that of natural forest (Nandakwang *et al.*, 2008; Phongchiewboon, 2008).

Little information is available on most tropical tree species about how they meet the preferred characteristics of framework species. So far, lists of good framework species have been published only for the wet tropics of Queensland, Australia, where the method originates (Goosem and Tucker, 1995), and for the seasonally dry tropics in northern Thailand, where the method is actively studied and promoted by Chiang Mai University's Forest Restoration Unit (FORRU) (Elliott *et al.*, 2003). FORRU has published detailed guidelines for studying and identifying framework species (FORRU, 2008; Box 12.2), and is carrying out research to identify framework species in Cambodia and other neighbouring countries.

Because the framework species method relies on natural recruitment of seedlings, its applicability depends on seed dispersers and dispersal distances of native tree species in the area. In tropical Asia, seed dispersal by mammals, large birds and bats is known to occur over distances of up to 10 km, while shorter distances from a few hundred metres to a few kilometres are probably more common (Corlett, 2009). Remnant forests must, therefore, be present within a few kilometres from the restoration site. The nearer the forest, the faster will be the recovery of species richness (FORRU, 2006). Although even scattered trees can act as seed sources, their seed may be largely inbred and thus of low quality for restoration (Blakesley, Hardwick and Elliott, 2002; Chapter 2). Subsequent enrichment planting is recommended if biodiversity recovery is not evident four to five years after tree planting (FORRU, 2006).

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Box 12.2.
Examples of tree species for attracting seed-dispersing animals in Thailand

Relatively few common fruit-eating animals are responsible for most seed dispersal between intact forest and restoration sites in northern Thailand. These include small to medium-sized birds, especially bulbuls, fruit bats (e.g. *Cynopterus* spp.) and certain medium-sized mammals, including civets, common wild pig, common barking deer and hog badger. These animals are equally at home both in forest and in deforested areas.

Tree species that are most likely to attract seed-dispersing animals to restoration sites produce small to medium-sized fruits within three years after planting. Such species indigenous in northern Thailand include *Callicarpa arborea, Castanopsis tribuloides, Eugenia grata, Ficus abellii, F. hispida, F. semicordata, F. subincisa, Glochidion kerrii, Heynea trijuga, Macaranga denticulata, Machilus kurzii, Prunus cerasoides* and *Rhus rhetsoides*. Some species also produce flowers with large quantities of nectar (e.g. *Erythrina subumbrans*). In general, local fig species (*Ficus* spp.) are good candidates for framework species because of their fruiting patterns and high survival even under unfavourable site conditions.

Some tree species can provide nesting sites for birds within five years after planting, further enhancing seed dispersal to the site. Such species in northern Thailand include *Alseodaphne andersonii, Balakata baccata, Bischofia javanica, Cinnamomum iners, Duabanga grandiflora, Erythrina subumbrans, Eugenia albiflora, Ficus glaberima, F. semicordata, F. subincisa, Helicia nilagirica, Hovenia dulcis, Phoebe lanceolata, Prunus cerasoides, Pterospermum grandiflorum, Quercus semiserrata, Rhus rhetsoides* and *Spondias axillaris*.

Source: Forest Restoration Research Unit, 2006. *How to plant a forest: the principles and practice of restoring tropical forests*. Chiang Mai, Thailand, Biology Department, Science Faculty, Chiang Mai University.

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12.3. Assisted natural regeneration

Evert Thomas

Bioversity International, Regional office for the Americas, Cali, Colombia

Approaches to ecological restoration of forest ecosystems depend strongly on the initial state of forest or land degradation, as well as the desired outcomes, time frame and financial constraints (Chazdon, 2008). In sites with low to intermediate levels of degradation where soils are generally intact (typically degraded [*Imperata*] grassland or shrub vegetation), natural regeneration of forest species is often sufficient to trigger the conversion to more productive forests with relatively little human intervention. This is what assisted natural regeneration (ANR) is all about: accelerate, rather than replace, natural successional processes by removing or reducing barriers to natural forest regeneration (Shono, Cadaweng and Durst, 2007). The method was originally proposed by Dalmaico (1986) and since then has gained considerable popularity around the world (FAO, 2003, 2012). One of the attractive characteristics of ANR is its cost-effectiveness compared with conventional reforestation methods, most of which have substantial costs associated with propagating, raising and planting seedlings (FAO, 2003; Shono, Cadaweng and Durst, 2007). As ANR involves less site preparation and nursery establishment, costs can often be as low as half to one tenth of those of conventional reforestation practice (FAC and DANIDA, 2005; FAO, 2012). Furthermore, ANR is very compatible with traditional systems of natural resource management, and easily understood by field staff (FAO, 2003). However, the method is generally labour-intensive, requiring nearly constant maintenance of selected forest areas for five to seven years to ensure establishment of desirable tree species (FAO, 2012). Hence, in order to obtain successful results it is crucial to involve local communities.

ANR aims at enhancing the establishment of secondary forests by protecting and nurturing the mother trees and their offspring already present in the area. This can be achieved by removing or reducing barriers to regeneration, such as soil degradation, competition with weedy species, and recurring disturbances (e.g. fire, grazing and wood harvesting) (Shono, Cadaweng and Durst, 2007). Particular care is given to liberating naturally regenerating seedlings or saplings from competition with undergrowth by weeding a circular area around them and to protecting them from fire and grazing (e.g. through active establishment of fuel breaks and fences, respectively). Where two or more seedlings or saplings are close to each other, the smaller, less healthy or less desirable one is removed and, where appropriate, transplanted to empty places in the restoration site (FAC and DANIDA, 2005). In some cases, fertilizer may be applied to promote the growth of existing seedlings or saplings.

ANR is most applicable in areas with remaining trees or patches of natural forest within a wider degraded landscape, as these trees provide propagation material or attract dispersal agents (birds, bats, mammals, etc). Five hundred to 800 wildings/ha is generally adequate to ensure establishment of a stable second-growth forest and eventual restoration of a dense forest cover (Shono, Cadaweng and Durst, 2007; FAO, 2012). Wildlife is an essential component in the restoration approach for its role in seed dispersal, and should therefore be protected (FAC and DANIDA, 2005). Precisely because of ANR's reliance on natural processes, it is especially effective in restoring and enhancing biological diversity and ecological processes (FAO, 2003). ANR is not to be recommended for ecological restoration in seriously degraded landscapes as it is likely that remaining isolated trees do not produce viable seeds or vigorous seedlings (FAC and DANIDA, 2005). Depending on the desired outcome, quantity and quality of natural regeneration (e.g. fewer than 500–800 naturally occurring wildlings per ha; FAO, 2012), time constraints and/or available financial resources, stands may need to be enriched with a variety of species, such as fast-growing, light-demanding species that create shade in the understorey and a habitat for late-successional species, orchard trees or commercial tree species. Thus, it is important to choose a wide variety of native species matched to different microclimatic conditions in the restoration area, including species that provide fruit for birds, bats and other animals that spread seed. Once nurse trees and existing woody species start casting appropriate shade the stand can be enriched with shade-tolerant (high-value) species (FAC and DANIDA, 2005). Hence, it is clear that ANR techniques are flexible and allow for the integration of various objectives, such as timber production, biodiversity recovery and cultivation of crops, fruit trees and non-timber forest products in restored forests (Shono, Cadaweng and Durst, 2007).

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For more information on assisted natural regeneration, see: http://www.fao.org/forestry/anr/en/

12.3.1. Assisted natural regeneration in China

Jiang Sannai[21]

Nearly 40 percent of China's surface area is seriously eroded, and more than one quarter of the land is covered with desert soils. Since the 1990s, the frequency of sand storms has increased, especially in northern China. Assisted natural regeneration (ANR) has played an important role in the country's effort to counter expanding environmental degradation. In China, ANR can be divided into two main categories: special ANR and general ANR. Special ANR is practised on cutover land with measures such as soil preparation conducted to improve site conditions for forest establishment. General ANR refers to more comprehensive regeneration and afforestation activities accompanied by artificial sowing, tending and other treatments. It is conducted on barren hills, wasteland, barren desert lands, cutover

[21] Based on Sannai (2003), published with permission.

lands, riverbanks with important ecological status, sandy regions damaged by wind and such like. The objective is to establish vegetative cover to protect the land. In regions where some natural sowing occurs, the area is closed to most forms of exploitation for a number of years (depending on local conditions). Use of the land is restricted or prohibited during this period to facilitate forest establishment from natural seed fall. Where natural sowing and natural regeneration are unlikely to occur without human assistance, the closed areas will be sown with tree and/or grass seeds from the air.

Closed areas are subject to both administrative and management measures. Three types of closure are distinguished in China:

1. Full-closure is adopted for a period of three to five years or eight to ten years (depending on local conditions) in regions such as remote mountains, upper reaches of rivers, water catchments of reservoirs, sites characterized by severe soil erosion, desert soil areas subject to wind damage and other regions where natural regeneration is difficult.
2. Semi-closure is practised in areas where some target tree species are growing well and where the percentage of forest cover is relatively high. Under semi-closure, strict protection is prescribed to protect the saplings and seedlings of target tree species. However, controlled cutting of fuelwood and grass may be allowed.
3. Full-closure and semi-closure are combined in regions where farmers are very poor and fuelwood is scarce. Full closure periods alternate with semi-closure periods. There are no fixed standards; the lengths of full or semi-closure period vary depending on the progress achieved in restoring vegetative cover.

Between 2001 and 2003, over 30 million hectares of forest was established through the closure system. Aerial sowing of tree or grass seeds was implemented in 931 counties of 26 provinces (autonomous regions and municipalities directly under the central government). Approximately 8.68 million hectares of forests were established through aerial seeding combined with closure. This accounted for 25 percent of the total artificial forests of China in 2003.

ANR has played a very important role in expanding forest resources, controlling soil erosion, retarding the process of desertification, improving the ecological environment and improving the living conditions of farmers.

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12.4. Post-fire passive restoration of Andean *Araucaria–Nothofagus* forests

Mauro E. González

Forest Ecology Laboratory, Faculty of Forest Science and Natural Resources, Universidad Austral de Chile and Center for Climate and Resilience Research.

In 2002 large and highly intense fires caused by lightning strikes affected vast areas of Andean *Araucaria–Nothofagus* forests (ANF) within several national parks and forest reserves in south-central Chile. The two worst-affected protected areas were Tolhuaca National Park and Malleco National Reserve, with over half of their combined total area burned (14 536 ha). Considering the scientific and cultural importance of *Araucaria araucana* (Molina) K. Koch, these events prompted the rapid development of plans to restore and evaluate the recovery of *Araucaria* forests in the two areas. Here, we present a restoration effort that used a passive approach.

Fire has been an intrinsic ecological process influencing the ANF, although its role has only recently been more fully understood (Burns, 1993; González, Veblen and Sibold, 2005, 2010; Quezada, 2008; Mundo, 2011). Fire regimes in the ANF are dominated by mixed-severity fires (low-severity surface fires and crown fires) that result in a range of fire effects and responses (González, Veblen and Sibold, 2005, 2010). Thus, fires typically result in a mosaic of vegetation burnt to varying degrees of severity (Peñaloza, 2007).

Fires, both naturally occurring and human-induced, have influenced the ANF over the past several millennia (Heusser, 1994). Since ancient times, native tribes of the Araucarian region (e.g. Pehuenche and Mapuche people) have used the area for activities such as hunting, grazing and the collection of *Araucaria* seeds (Aagesen, 1998; Tacón, 1999; Bengoa, 2000). Fire was typically used by native tribes for hunting guanaco (*Lama guanicoe*; Veblen and Lorenz, 1988), and possibly to clear undergrowth vegetation to facilitate the collection of *Araucaria* seeds. With the introduction of domestic livestock (by the late 1500s), fires were also used to clear travel corridors and manipulate forage. After the arrival of Euro-Chilean settlers to the Araucarian region (after 1882) human-induced fires increased dramatically. Fire was used as the main tool to clear forests for agriculture and cattle grazing and also to improve pasture quality in high-altitude woodland valleys covered usually with open woodlands of *Araucaria–Nothofagus antarctica* (G. Forster) Oerst. In sum, humans have had a significant impact on the historical fire regime of these ecosystems (Gonzaléz, 2005; González, Veblen and Sibold, 2005; Quezada, 2008).

Post-fire early secondary development as passive restoration of Araucaria–Nothofagus *forests*

Although most details of post-fire *Araucaria–Nothofagus* forest recovery still are not completely understood, secondary succession is recognized as an important process in ecological restoration. Research on early secondary succession provides basic information on key ecological processes and species' responses to enhance forest restoration activities (i.e. methods and procedures) and ecosystem integrity. Passive restoration – the recovery of forest by natural regeneration after fire – has been used as an initial step before active restoration is implemented. Where the natural forest response achieves the desired processes, functions, structure and composition, restoration may rely mostly on natural recovery. The present case study is intended to illustrate early post-fire ANF recovery in terms of recruitment of tree seedlings and understorey species under the effect of two different fire severities, moderate and high.

Study area and sampling design

This study was carried out in subalpine temperate old-growth *Araucaria araucana–Nothofagus pumilio* (P. et E.) Krasser forests (c. 1200 m above sea level) within the Tolhuaca National Park (38°12'S and 71°45'W; Figure 12.7). This area – especially high-altitude open woodland intermixed with grassland – was historically used for summer cattle ranching both by Native Americans (c. 1700–1900) and later (1900–1960) by early settlers, who established nearby the protected areas. At lower elevations (less than 900 m above sea level) *Nothofagus* forests were selectively harvested between 1940 and 1960.

During the summer the Tolhuaca National Park is visited by many people for camping, fishing and trekking. These recreational activities, especially trekking, have been negatively affected by the 2002 fire because of the danger from fallen dead trees.

After the 2002 fire, we established six permanent plots of 1000 m^2 to evaluate vegetation recovery in areas affected by mid- and high-severity fire. Moderate fire severity killed 40 percent to 60 percent of trees and consumed most of the undergrowth. High fire severity killed more than 95 percent of trees and the undergrowth was completely consumed. In each plot we measured the diameter at breast height (dbh) of all live and dead trees with a dbh greater than 5 cm. Vegetation response was evaluated in 30 subplots of 1 m^2 systematically laid out in each of the six plots, where we counted the number of saplings (less than 5 cm dbh and greater than 2 m height) and seedlings of tree species (less than 2 m height), and estimated the abundance-dominance of undergrowth species using Braun-

Figure 12.7.
Location of Tolhuaca National Park in the province of Malleco, Araucanía region, Chile. Circle indicates the study site, classified based on severity of fire.

Figure 12.8.
Post-fire recruitment of seedlings in *Araucaria–Nothofagus* forests burned by fires of medium and high severity

Blanquet cover-class values. Importance value for each species was determined by calculating the sum of the relative frequency and relative cover.

Tree seedling regeneration under different fire severities

Fire severity significantly influenced tree seedling recruitment in the ANF (Figure 12.8). Recruitment of *Nothofagus* species was greatest following fire of moderate severity, which left remnant trees alive. Seedlings of *N. dombeyi* (Mirb.) Oerst. and *N. pumilio* originated from wind-dispersed seeds. Seedlings of *N. nervosa* (Phil.) Krasser originated from basal resprouts and also from wind-dispersed seeds. By contrast, recruitment of medium-sized pioneer and opportunistic trees such as *Embothrium coccineum* J.R. et G. Forster and *Lomatia hirsuta* Diels ex Macbr. (Proteaceae family) was greater following fire of high severity. Recruitment of these species originated through resprouting of basal buds (when individuals were present in the former, more open stands) and from wind-dispersed seeds. Seedlings of *Araucaria araucana* established either from gravity-dispersed seeds – seeds protected inside the cones of female trees (González *et al*., 2006) – or resprouts from basal buds of burned juvenile trees (Figure 12.9). Seedlings of *N. pumilio* were unable to establish following high-severity fire. This fire-sensitive species (González, Veblen and Sibold, 2005) is an important component of the original forest stand. Given that it is dependent on seeds for recruitment and its seeds have only a limited range of dispersion, remnant trees are key for its successful re-establishment.

Bamboo (*Chusquea culeou* Desv.) colonized the sites more rapidly than any other understorey species (Figure 12.10). This species reached importance values (IV) of 32 percent following high-severity fire and 62 percent following moderate-severity fire. Moderate-severity fire lightly burned some patches of bamboo, causing minor damage to the rhizome system and allowing a rapid response. Other understorey species reached higher IVs following high-severity fire.

Figure 12.9.
a) Seedlings of *Araucaria araucana* from seeds that survived high temperature inside the cones
b) basal resprout of a relatively juvenile individual of *Araucaria araucana*

a)

b)

Figure 12.10.
Araucaria–Nothofagus stand affected by a high severity fire. Note the dense resprouting of bamboo culms (*Chusquea culeou*)

These included *Muelhenbeckia hastulata* (J.E. Sm.) Johnst., *Alstroemeria aurea* R. Graham, *Dioscorea brachybotrya* Poepp., *Ribes magellanicum* Poir and *Gaulteria phillyreifolia* (Pers.) Sleumer. The cosmopolitan species *Senecio vulgaris* L. (Asteraceae) colonized the site via wind-dispersed seeds, forming relatively dense patches, especially in sites with a lower cover of *Chusquea culeou* affected by high-severity fire. Other invasive weeds (e.g. *Hypochoeris radicata* L. and *Cirsium vulgare* (Savi) Ten.) were brought to the sites by cattle which occasionally grazed the area.

Main conclusions

Fires burn forest stands with different severities, providing various opportunities for species recruitment. Fire severity influences the amount of organic material destroyed and hence the amount and types of biological material that remain after fire. The number of live trees and reproductive structures remaining below ground has an important influence on post-fire regeneration of woody species. The recruitment of the obligate seeders *Nothofagus dombeyi* and *N. pumilio* was generally low following high-severity fire. The almost complete eradication of the adult population of both fire-sensitive species contributed to low seed availability post-fire and hence restricted opportunities for seedling establishment in the post-fire environment. In contrast, *Araucaria araucana* and *N. nervosa*, which are capable of resprouting after fire, were able to establish following both medium- and high-severity fires. Seedlings of *Araucaria* established under remnant female trees.

Even though the impact of the presence of domestic livestock has not been evaluated, observations indicate that livestock can have an important influence on the process of forest recovery. In the early stages of recovery, trampling, grazing and browsing have significant detrimental effects on tree seedling survival and growth, especially for the most palatable species (i.e. all *Nothofagus* species). Moreover, the combined effect of severe fires and cattle would favour the dominance of shrub species (e.g. *Chusquea culeou*; Raffaelle *et al.*, 2011). The high tree mortality and burning of the undergrowth seem to promote weed growth and facilitate (or attract) the presence of cattle.

These preliminary findings indicate some general considerations and recommendations to enhance a passive restoration approach. First, it is important to recognize that fire severity influences stand composition and structure following the fire, which may result in different successional pathways in the forest community. That could be the case especially for severely burned stands where *Chusquea culeou*, an undergrowth species, can outcompete the relatively poor tree seedling recruitments because of its strong ability to rapidly resprout from its rhizome system, which covers the site at high density. Second, it is important to evaluate the responses of the dominant tree species, especially for fire-sensitive species, to the new (abiotic and biotic) conditions following fires of different severities. Although *N. dombeyi* and *N. pumilio* typically establish with high density after stand-replacing fires (Mera, 2009; González, Veblen and Sibold, 2010), a very severe fire can hamper or delay the successful establishment of the main canopy species. Under this scenario, active restoration could be implemented by supplementary (enrichment) planting of tree seedlings. Third, the post-fire environment of strong light, bare soil and lack of groundcover competition provides a temporary opportunity for abundant recruitment of weed species. Therefore, monitoring and controlling exotic weeds and domestic cattle is an important measure to favour successful passive restoration of the burned forests. Passive restoration together with a little active assistance could be an effective way to restore ecosystem function, integrity (community composition and structure) and sustainability (resistance to disturbance and resilience).

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12.5. Carrifran Wildwood: using palaeoecological knowledge for restoration of original vegetation

Philip Ashmole

Borders Forest Trust, Monteviot Nurseries, Ancrum, Jedburgh, United Kingdom

Carrifran valley in the Southern Uplands of Scotland was denuded when Borders Forest Trust (BFT) purchased it by public subscription in 2000 and commenced restoration of a "wildwood." The vision of the Wildwood Group (a somewhat devolved element within BFT) was that by removing negative anthropogenic factors and initiating woodland development by planting, it might be possible to restore a broadleaved forest and moorland ecosystem similar to that which existed in the 650 ha valley about 6000 years ago. This was a time when the primary forest of Scotland probably reached its greatest extent and diversity, following immigration of all major tree types (Tipping, 1994). The subsequent loss of natural forest was caused primarily by human activity. Climate and soils also changed to some extent, but the large altitudinal range of the site, coupled with the great variety in aspect, slope and substrate, led to an expectation that most of the original species of fungi, plants and animals would still encounter suitable conditions somewhere on the site, thus enabling restoration of nearly original/natural woodland (Peterken, 1996, 1998).

Having secured Carrifran, the Wildwood Group organized a discussion meeting to provide a basis for restoration of broadleaved native woodland on this site and elsewhere in southern Scotland (Newton, 1998; Newton and Ashmole, 1998). Previously, attention of Scottish environmentalists had focused mainly on the Highlands and especially on native pinewoods, where different considerations might apply.

Choice of appropriate woody species for planting on site was of immediate concern. Palaeoecology can supply a record, although inevitably incomplete, of the taxa that have occupied a site at various times in the past. At Carrifran a core through the peat at 620 m in Rotten Bottom provided palynological data extending back to the early Holocene; this was supplemented by data from two other sites within 5 km of Carrifran (Tipping, 1998). This information was used as one basis for a list of tree and shrub species considered native to Carrifran valley (Newton and Ashmole, 1999).

The uncertainties associated with pollen analysis make it desirable to supplement the palaeoecological record with other types of information. For instance, existing ancient woodlands can demonstrate the suitability of a region for those tree and shrub species that occur within them. In the Southern Uplands, however, surviving ancient woodlands are rare, isolated, small and often linear (Badenoch, 1994) and it has been argued that their use as a template for ecological restoration of a denuded site might lead to establishment of a woodland that was a degenerate and species-impoverished reflection of the past (Tipping, 1998).

This danger can be countered to some extent by use of the national vegetation classification (NVC), which is based on information from a wider range of British sites (Rodwell, 1991). This makes it possible to use existing open-ground vegetation as a predictor of the appropriate composition for native woodland to be established on the site (Rodwell and Patterson, 1994; Averis, 1998). Caution is required because the NVC framework is based on existing British vegetation types, but most woodlands and other natural habitats have been subject to a variety of human influences

Figure 12.11.
Carrifran valley

over many centuries. These include selective management for useful species and lack of protection from grazing and browsing by domesticated herbivores (Smout, MacDonald and Watson, 2005).

NVC analysis was used at Carrifran, however, to decide on the most appropriate composition for woodland in the various parts of the site, which differ in altitude, aspect, slope, soil and moisture. Additional insight was obtained by the use of ecological site classification (ESC) analysis developed by the Forestry Commission, which is based on assessment of three principal factors: climate, soil moisture and soil nutrient regime (Pyatt and Suárez, 1997). At Carrifran this analysis was particularly influential in emphasizing the role of juniper (*Juniperus communis* L.) on the high plateau around the rim of the valley.

Choice of appropriate species is a crucial first step, but must be linked with a strategy for obtaining appropriate seeds or cuttings to establish on site. Genetic advice was clear: the aim should be creation of a dynamic and expanding woodland resource with the capacity to evolve in the future and respond to environmental change (Ennos, 1998). To achieve this aim, planting stock should be sourced from relict ancient woods near to the restoration site and with conditions matching it as closely as possible. However, populations in isolated and small woodland remnants may have low genetic variability and differ from one another in genetic composition because of historically low population sizes. In order to maximize genetic diversity in the new population it is necessary to collect propagation material from numerous individuals and from several different suitable sites.

Some of the natural Scottish tree populations listed by Wilson, Malcolm and Rook (2000) seemed appropriate as sources of seed for Carrifran, but other batches of seed were obtained from woodland fragments that appeared ancient, even though there was no documentary evidence of their status. For some species there were particular problems. For instance, in sessile oak (*Quercus petraea* (Mattuschka) Liebl.), the desired oak species for Carrifran, hybridization between native trees and planted pedunculate oak (*Quercus robur* L.) made most local populations suspect. In the early years of planting some acorns were obtained from Cumbria, but doubts about the status of the woods there led to a switch to Galloway as the main source. However, a few remote woods in Cumbria are probably ancient and may contain trees adapted to high altitudes, so a special effort was made to collect seed from them for planting in the high parts of Carrifran.

Aspen (*Populus tremula* L.) was also a problematic species, since it is now represented in southern Scotland only by widely scattered, small and often clonal stands. Yet it was an early colonizer of Scotland and may have been a significant component of pristine native woodland on a wide variety of soil types and from sea level almost to treeline (Quelch, 2002). By collecting root cuttings for propagation from about 20 different stands and planting the progeny in many parts of Carrifran, it was hoped that a representation of aspen similar to that in the natural woodland would eventually be achieved.

Now that many trees are more than a decade old it has become obvious that the rate of tree growth decreases markedly between the floor of the valley, at around 250 m above sea level, and the upper limit of the main planting at 450–550 m above sea level, which may be around the timberline (Ashmole, 2006). In recent years the Wildwood Group has paid special attention to planting above this level, in an attempt to restore treeline woodland and montane scrub, habitats that have been almost entirely lost from Scotland (Hester, 1995; Gilbert, Horsfield and Thompson, 1997; Ashmole, 2006; Chalmers and Ashmole, 2007).

Field observations and data on accumulated temperature and relative windiness indicated that the land above about 750 m above sea level would not support woodland or scrub and that there would be a natural transition to montane moss-heath, which would extend to the windswept summit of White Coomb (821 m), the fourth highest peak in the south of Scotland (Hale, Quine and Suárez, 1998; Adair, 2005). However, a high hanging valley at Carrifran provided an opportunity to attempt establishment of montane scrub between 600 m and 750 m above sea level. Some 12 500 shrubs were planted between 2007 and 2012, and although mortality is significant and growth very slow, scrub vegetation is becoming established. Emphasis has been on juniper, sourced from the highest available natural populations in the area, and on downy willow (*Salix lapponum* L.), which in Britain has relict populations in only three localities south of the Scottish Highlands, one of them only 1 km from Carrifran.

Thirteen years after the start of the restoration work at Carrifran, over half a million trees have been planted and about 300 ha of native woodland are well established in the lower half of the valley. Ground vegetation is changing rapidly and woodland animal species are colonizing the newly created habitats (Ashmole and Ashmole, 2009). In years to come, as active management of Carrifran is reduced and natural processes come into play, it is hoped that this catchment in the heart of the Southern Uplands can provide an exemplar of a functioning ecosystem similar to that which would have been present in the absence of destructive human intervention during the second half of the Holocene.

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12.6. The Xingu Seed Network and mechanized direct seeding

Eduardo Malta Campos Filho, Rodrigo G. P. Junqueira, Osvaldo L. de Sousa, Luciano L. Eichholz, Cassiano C. Marmet, José Nicola M. N. da Costa, Bruna D. Ferreira, Heber Q. Alves and André J. A. Villas-Bôas

Instituto Socioambiental, Xingu, Mato Grosso, Brazil

The Xingu River flows from the tropical savannah of central Mato Grosso north to the Amazon. With a length of nearly 2000 km, the area it drains boasts extensive water resources, biodiversity and human diversity. The 24 culturally distinct indigenous peoples of Xingu have conserved most of the native vegetation in their territories along the rivers, but settlers who arrived during the last 40 years have deforested much of the area on the headwaters of those rivers to create fields for growing soybean and pastures for cattle ranching. Although prohibited by the Brazilian Forest Code, the deforestation of 300 000 hectares of the riparian zone has jeopardized water quality and regulation of water flow, as well as harming the health of people who, for centuries, have depended on the river for water, food and other services.

Since a meeting in 2004, Instituto Socioambiental (ISA; www.socioambiental.org.br) has brought together the region's stakeholders into a campaign called "Y Ikatu Xingu" (Save the Good Water of Xingu, in the language of the Kamaiura Indians) with three principal components: (i) forest restoration; (ii) education and communication; and (iii) regional cooperation between non-governmental organizations (NGOs), communities and policy-makers.[22]

In 2006, ISA and its partners started educational programmes with teachers, students, extension agents and officials, while farmers were offered technical assistance, material and financial support (mostly seeds and fences) for the restoration of riparian zones. The objectives of the forest restoration work included protection of water resources, fruit production, timber production, carbon sequestration and legal compliance with the Forest Code, and thus addressed the needs of a wide range of farmers.

Each restoration project has been aligned with farmers' knowledge and ideas to ensure it does not require major changes from farmers' existing practices. Indigenous peoples stated that trees must be planted by direct seeding, so that roots develop deeper and the trees can survive drought. As a result, direct seeding with common agricultural machinery has been a much better accepted and effective option than planting seedlings. Farmers use the machines and knowledge used for growing soybeans, maize or grasses or for spreading fertilizers and limestone to plant native trees.

Direct seeding also proved to cost less than planting seedlings (approximately US$2000/ha, compared with US$5000/ha) and to be more practical, since seeds are easier to carry and to plant. To plant one hectare, approximately 60 kilos of seeds of native trees (200 000 seeds) are mixed with 100 000 seeds of annual and subperennial legumes and sand, in a mixture called *muvuca*. The legumes help to create a multilayer vegetation, reducing niches for invasive grasses. Their root systems can contribute to soil aeration and decompaction, enhancing water absorption. Their ability to fix nitrogen and their intense leaf fall contribute to enhancing nutrient cycling and soil fertility. Their flowers and fruits attract fauna

[22] http://www.yikatuxingu.org.br

and can be sold. However, if they grow too densely, they can shade out the tree seedlings, slowing tree growth. If this occurs, manual or chemical weeding or thinning will be necessary.

Ninety-one of the native tree species planted have germinated and survived droughts of up to six months without irrigation. Tree populations of between 2500 and 32 250 trees/ha have established on the reseeded areas. The oldest planted area is six years old and has a mean density of 7250 trees/ha, greater than the 1666 trees/ha conventionally used when planting seedlings. Natural thinning seems to occur over time as a result of ant herbivory and other mortality factors.

The campaign restored 2565 ha of riparian forest at 238 sites. The demand for seeds of indigenous tree species rose dramatically and was met by the creation of the Xingu Seed Network,[23] formed by 300 indigenous people, small landholders and peasants. Between 2006 and 2012 the Network produced and sold 71 tonnes of seeds of 214 indigenous species and earned almost US$400 000 from the environment they have preserved.

Figure 12.12.
Women in the Panará Indigenous Territory gathering seeds from the Amazon forest, Guarantã do Norte, MT, Brazil

Figure 12.13.
Kaiabi and Yudja people teaching techniques for seed gathering in the forest at São José do Xingu, MT, Brazil

[23] http://www.sementesdoxingu.org.br

Figure 12.14.
Preparing *muvuca*, a mixture of seeds of native trees, fast-growing legumes and sand for direct seeding

Figure 12.15.
Mechanized direct seeding: using machines designed for spreading fertilizers (left) and sowing soybeans (right), Mato Grosso State, Brazil

Figure 12.16.
Restoration plot five months after being planted with *muvuca* containing pigeon peas, jack beans, maize and tree seed, Canarana, MT, Brazil

Figure 12.17.
Same area shown in Figure 12.16 after three years, with the trees forming the new canopy, the last pigeon peas dying, and jack beans and maize already out of the ecosystem, Canarana, MT, Brazil

Network annual meetings discuss ecological knowledge about indigenous tree species and techniques for collecting and cleaning seeds and set prices for seed of each species. Seed gatherers are organized in local groups, and each group is represented by one of its members. Groups make lists of what species they can collect each year and in what quantity. Based on these lists, farmers and NGOs order what they want to buy. A microcredit fund allows seed-gatherers to invest in their activities.

Seeds are stored in four storage facilities called "seed houses," which are equipped with air-conditioning and a dehumidifier. Each seed lot comes with information regarding who collected it, where it was collected, type of vegetation, name of the species, number of parent trees and date of collection. Lots are assigned a control number at the seed house and 100 seeds are taken out for viability tests. Seed quality is checked at least three times: when seeds are selected in the forest, during cleaning and drying and at the seed houses. Seed technology is still a great challenge for many species, but a lot has been developed by the Network, filling information gaps.

Results and learning are disseminated through field-day demonstrations, practical courses, lectures, workshops, videos, television, magazines, newspapers, interchange expeditions, school activities and demonstration areas.

Focusing on restoration of riparian zones in a drainage basin has proved successful from a practical point of view because it addresses water conservation and quality issues and thus can get people engaged. From a wider forest conservation point of view the approach also serves to connect fragmented patches of forests across the landscape and thus promotes gene flow and diversity at the landscape level.

Chapter 13

Approaches including production objectives

13.1. Analogue forestry as an approach for restoration and ecosystem production

Carlos Navarro[1] and Orlidia Hechavarria Kindelan[2]

[1] Coordinating Committee, Latin American Forest Genetic Resources Network
[2] Agroforestry Research Institute, Cuba

"Analogue forestry" is a restoration approach that aims to develop production- or conservation-oriented forest systems in degraded forest areas by drawing on knowledge and observations about local climax vegetation (Senanayake and Jack, 1998). The approach is based on the structure and composition of Sri Lankan and Indonesian forests and home gardens, which are small plots of highly productive land located near houses in traditional rural communities (Senanayake and Jack, 1998; Gamboa and Criollo, 2011). These gardens maintain a wide diversity of trees, shrubs and herbs in a manner similar to a forest, and represent an important part of the traditional knowledge of farmers.

Forest gardens also serve as a safety net. In Indonesia rice is the staple food for most people. Other crops such as cassava (*Manihot esculenta* Crantz), taro (*Colocasia esculenta* (L.) Schott) and sweet potato (*Ipomoea batatas* (L.) Lam.) are grown in forest gardens but rarely consumed by humans. These crops, which are considered "food of the poor," are often used as feed for domestic pigs. However, in times of difficulty, when the rice harvest fails or when rice stocks are exhausted just before the next harvest, people use cassava, taro and sweet potato and other crops from their gardens as emergency foods (Brodbeck, Hapla and Mitlöhner, 2003). The same is true for fruit species in Costa Rica, where part of the production in gardens is not collected when farmers are busy tending their coffee or cocoa crops. These fruits are, however, important in times of crisis. These garden sites, locally called *solares*, also provide medicinal plants, basic foods and fruits, have an ornamental value and benefit the environment.

Analogue forestry attempts to both increase biodiversity and improve the well-being of local communities by creating enhanced and diversified production systems, valuing people's own resources and promoting respect for local values and traditions. It uses a wide range of crops and hence reduces risks to farmers of being dependent on a single product. The approach aims to recreate ecosystems based on the structure and ecological functions of the original vegetation, facilitating the spread of many species from the original forest. It is used to accelerate restoration in highly degraded areas, especially when there is little natural gene flow from the surrounding areas. Analogue forestry also allows use of exotic species that are similar in structure and function to native species if the native species has disappeared due to fragmentation or habitat loss.

Analogue forestry builds on 12 guiding principles

1. **Observe and record**: observation and recording of the structure and physiognomy of the original forest and vegetation and the soil conditions of the site to be restored.
2. **Understand and evaluate**: information on vegetation, soils, wind directions, water flow, hedges or artificial fences, etc. is collected both in the natural forest and the area to be restored and is analysed.
3. **Know your land**: gather all available information and knowledge on the soil and biodiversity conditions of the land.
4. **Identify levels of yield**: determine potential crop yields in the target area; this should be done for all possible products (e.g. cocoa, coffee, vanilla, timber).
5. **Map flow and reservoir systems** (existing and potential): prepare maps of water flows, water tanks and others components of the hydrological system.
6. **Reduce ratio of external energy in production**: avoid using external inputs when the necessary inputs can be sourced locally.
7. **Be guided by the landscape and the needs of the neighbours**: look at the site as part of a larger unit to ensure an integrated approach to site restoration.
8. **Follow ecological succession**: if the system is degraded, plant pioneer species to improve soil conditions for other, more-demanding species.
9. **Make use of ecological processes**: when the system has been damaged by erosion or overgrazing by livestock, start with pioneer species to improve soil conditions to allow the site to support a climax ecosystem at a later stage.
10. **Value biodiversity**: combine as many climax forest species as possible, although it is sometimes difficult to obtain germplasm of all the species desired.
11. **Respect maturity**: mature ecosystems are often more productive than early-state systems in terms of biomass production and ecosystem services, and are especially important for photosynthesis and carbon and water cycles.
12. **Respond creatively**: create a system where species associations and diversity of components help to control pests and diseases in an ecosystem approach.

Contribution of analogue forestry to forest regeneration

Under natural regeneration a forest may take 40 to 60 years to achieve something approaching its original state, with a return of 60–70 percent of the original flora and fauna. Analogue forestry helps to reduce this period by accelerating ecological succession. It follows a natural pattern throughout the restoration process. Starting with pioneer species, and by promoting ecological succession, analogue forestry modifies the structure of the forest canopy and soil quality to allow the site to support a system of climax vegetation similar to natural forest of the area. Pioneer species facilitate restoration by helping improve soil conditions for more demanding and late-successional species.

In contrast with many other restoration techniques, analogue forestry does not focus on only woody species, because at least 90–95 percent of the biological diversity of forest plants in many ecosystems is in non-arboreal components (shrubs, grasses, epiphytes and lianas; RIFA, 2005). The idea is to create a system in which various species, products and plant combinations that can help controlling pests and diseases are considered in an integrated manner.

The soil at some restoration sites is not able to support a climax ecosystem, and needs to be modified. In newly formed soils (e.g. those that are the product of volcanic eruptions or sedimentary soil processes such as flooding), the prevailing habitat conditions may impede the development an ecosystem that is similar to the natural vegetation of the area. On such sites, the first step is to study the surrounding pioneer vegetation and natural forests and describe their physiognomy, structure, species composition and interactions, both in terms of density and in their vertical or horizontal spatial arrangements. The next step is to replicate this vegetation in the new areas to assist natural regeneration.

Genetic diversity conservation and analogue forestry can be combined to produce a better environment and more resilient ecosystems. Analogue forestry can contribute to the conservation of genetic diversity by:

- providing space for diversity conservation;
- establishing spatial arrangements that encourage gene flow and increase connectivity between patches of forests;
- preserving ecological relationships among species; and
- creating demonstration plots for environmental education.

In turn, analogue forestry benefits from genetic diversity, for example, through increased resistance to pests and diseases, better local adaptation, adaptation to climate change and diversity of products.

Methodology

During the initial stage of a restoration project, the natural forest surrounding the area to be restored must be studied to determine the ecosystem to be re-established on the degraded area. The structure of the ecosystem is important to maintain the ecological relationships between pollinators, herbivores and nutrient cycling. To begin with, the number of canopy layers or strata should be determined, and woody plants in each of these identified. The height, coverage, consistency and leaf size of the prominent or dominant species in each stratum should be determined. Next, the different growth forms (e.g. climbers, small palm species or herbs) are described, including their average height or height range and coverage. Information must be gathered on the ecological roles and human uses of the species, especially where the restored system will provide staple or cash crops. Species are selected for the restoration of the various vegetation layers based on their growth height, their uses and the characteristics of their seeds, among other factors.

One tool that differentiates analogue forestry from other restoration techniques is the use of a physiognomic formula. It allows visualizing a model for the restoration process as a codified description of the structure of the tree and non-tree components of the vegetation found in the area of interest. In the formula, each stratum is described by a specific code, followed by a description of special growth categories. In highly diverse landscapes, the physiognomic formulas of the vegetation are complex, as they include all strata and life forms of the forest vegetation. Some of the criteria upon which such assessments are based are soil quality, biodiversity and vegetation structure.

The application of this approach is described in more detail in the following case studies from Costa Rica and Cuba.

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13.1.1. Restoring forest for food and vanilla production under *Erythrina* and *Gliricidia* trees in Costa Rica using the analogue forestry method

Carlos Navarro

Coordinating Committee, Latin American Forest Genetic Resources Network

An ecological assessment of the forest vegetation of Los Espaveles primary forests at the Centro Agronómico Tropical de Investigación y Enseñanza (CATIE), Turrialba, Costa Rica, identified the following strata, starting for the tallest trees:

- First (topmost) layer (V8): trees of more than 35 m tall; evergreen broadleaf species; sparse canopy cover of 6–25 percent of the forest area.
- Second layer (V7): woody plants; evergreen broadleaf species, height 20–35 m; patchy canopy cover (25–50 percent of forest area).
- Third layer (V6): woody plants; evergreen broadleaf species, height 10–20 m; sparse canopy cover.
- Fourth layer (V5): woody plants; evergreen broadleaf species, height 5–10 m; sparse cover.
- Fifth layer (V4): evergreen broadleaf species, height 2–5 m; patchy cover (1–6 percent).
- Sixth layer (V3): evergreen broadleaf species, height 0.5–2 m; sparse cover.
- Seventh layer (V2): evergreen broadleaf species, height 0.1–0.5 m; patchy cover (1–6 percent).
- Eighth (lowermost) layer (V1): seedlings of evergreen broadleaf species, length <2.5 cm, height <0.1 m; very low cover (less than 1 percent).

In addition, the following growth forms were identified:

- Climbers in canopy layers above 35 m (T8); almost absent
- Epiphytes in 10–20-m layer (E6); almost absent
- Palm species in 5–10-m layer (P5): sparse cover
- Bananas in 2–5-m layer (R4); sparse cover
- Ferns in 0.1–0.5-m layer (F2); sporadic cover
- Herbs in 0.1–0.5 m layer (H2); sporadic cover
- Lichens and mosses in <0.1 m layer (L1); sporadic cover.

The corresponding formula of this forest type is:

V8r, V7p, V6r, V5r, V4e, V3r, V2e, V1a; T8a; E6a; P5r; R4r; F2e; H2e; L1e

where the first letter (capital) indicates the growth form (V=evergreen broadleaf), the number indicates the canopy layer and the following lowercase letter indicates extent of cover (r=rare [6–25 percent], p=patchy [25–50 percent], e=sporadic [1–6 percent] and a=almost absent [<1 percent]). For non-woody plants the code also includes information on plant biology (E=epiphytes, P=palm trees, R=herbaceous [banana], F=ferns, H=herbs and L=lichens and mosses).

The physiognomic formula of the area to be restored is then constructed to allow comparison with the formula of the original forest followed by restoration of the layers that are missing or underrepresented:

- V6: evergreen woody plants, height 10–20 m; patchy cover
- V5: evergreen broadleaf plants with hard leaves (d), height 5–10 m; sparse cover
- V4: evergreen plants with hard leaves (d), height 2–5 m; scarce cover
- Special growth forms:
- Palm species, height 0.5 –2 m; patchy cover (P3)
- Epiphytes, height 5–20 m with soft leaves (s); almost absent (E6)
- Graminoids, height <0.1 m with soft leaves (s); continuous cover (c) (G1)

The resulting physiognomic formula of the area to be restored is

- V6esm, V5rdm, V4rdg; P3e; G1csn; E6-5sma

where m, g and n indicate leaf size.

As can be seen, the formula in the restoration area has fewer elements. The differences between the formulae for the natural vegetation and the area to be restored guide the restoration activities. In consequence, it is important to have detailed knowledge of species that can be used in the area, their potential ecological and economic uses and their stratum in the analogue layer of the forest.

CATIE has developed a demonstration site in Turrialba, Costa Rica, where plants for food production and vanilla were planted in association with *Erythrina* and *Gliricidia* trees. The goal of the project was to develop a productive site with a diverse group of species that: (i) have multiple uses; (ii) provide stability through the availability of products year round; and (iii) allow farmers to respond more flexibly to market fluctuations. An additional aim of the project was to simultaneously conserve diversity, provide wildlife with food sources and support a diversified production system.

In addition to plants that established through natural regeneration, the project planted other native and exotic species with: (i) medicinal value; (ii) importance for household consumption; (iii) commercialization potential; and (iv) potential to generate ecosystem services. Many of the species used provide multiple services including fruit, wood and erosion control. These include *Erythrina* sp., *Gliricidia sepium* (Jacq.) Kunth ex Walp., soursop (*Annona* sp.), *Caimito* (*Pouteria caimito* Radlk.), *Guaba* (*Inga* sp.), guava (*Psidium* sp.), *Zapote* (*Pouteria* sp.) and peach palm (*Bactris gasipaes* Kunth). Species planted for wood production, either for commercial timber or for use on farm, included mahogany (*Swietenia macrophylla* King), cedar (*Cedrela odorata* L.), nance (*Byrsonima crassifolia* (L.) Kunth) and two varieties of naranjilla (*Solanum quitoense* Lam.), among others. Medicinal plants included mint (*Mentha* sp.), rue (*Ruta* sp.) and oregano (*Origanum* sp.); several of the other species listed above also have medicinal properties.

13.1.2. Restoration of ecosystems on saline soils in Eastern Cuba using the analogue forestry method

Orlidia Hechavarria Kindelan

Agroforestry Research Institute, Cuba

At present, large areas of agricultural land in Cuba are degraded by salinization as a consequence of poor soil management associated with sugar-cane production. This degradation could be reversed through reforestation and conservation measures. Accumulating humus neutralizes the toxic effects of salinization and vegetation cover helps to maintain moisture in the top soil, which in turn impedes the concentration and crystallization of salts.

Study area

Numerous new farms were established in the vicinities of the communities of Cecilia, Sombrilla and Paraguay in 2000 as part of the national forestry farms plan which aimed at encouraging farm families to live in and reforest degraded wooded areas. The three communities are completely dependent on the local sugar mill for income, and relationships between them and the new farmer communities are not very good. There is no original forest left in the area, and the landscape is fragmented by cultivation of sugar cane. The farmers live on their farms with their children, their average age being between 33 and 35 years. The majority of them work on the farms as forestry workers. Women mostly take care of the household and participate in agricultural work. The study reported here took place near the community of Paraguay.

The original vegetation of the study area was mainly composed of xerophytic evergreen shrubs and thorny leguminous trees. On the degraded areas this vegetation has replaced by degraded secondary plant communities (Figure 13.1).

Mean annual precipitation fluctuates around 600 mm, with rainfall being concentrated from May to October. Mean annual temperature is approximately 26 °C. Soil in the area is Fluvisol of moderate depth (20–50 cm), fairly saline, with almost flat topography and little erosion (Sánchez *et al.*, 2008).

Methods

Approximately 75 soil samples were collected at depths of 0–20, 20–40, 40–60, 60–80 and 80–100 cm to assess the relation between salinity and the occurrence of indicator plants. At a depth of 0–20 cm, the soil salinity ranged from very low (EC_e 1.08 dS/m) to very high (EC_e 5.28 dS/m), with most samples being highly saline. Deeper soil layers were evaluated as excessively saline (EC_e 4.00–13.01 dS/m).The aquifer was also saline. The drainage canal was clogged, and the soil contained little organic matter. As a result of these unfavourable soil conditions the survival of tree species planted by local farmers was low.

In an effort to remedy this situation, the Cuban Agroforestry Research Institute, in coordination with the International Analog Forestry Network and the Falls Brook Centre, Canada, commenced a cooperative project to restore the vegetation of the xerophytic corridor of Guantanamo Valley using the analogue forestry approach. The project commenced in 2008 with the elaboration of maps of the farms and landscapes. This initial exercise showed a total farmed area of 338 ha. Species already occurring on site and other wild species capable of reducing the impact of degradation were identified from surveys, bibliographic information, interviews with community members and experts and results from previous work and selected for planting in the initial restoration phase. Once the ecosystem showed signs of stabilization, other exotic and native species were introduced.

Figure 13.1.
Degraded farmlands near the community of Paraguay before restoration work had begun (2008)

Medium-growth tree species already present on site before initiation of the project included *Casuarina equisetifolia* Forst, wild tamarind (*Lysiloma latisiliquum* (L.) Benth) and *Caesalpinea violaceae* (Mill.) Standl. During the first phases of the restoration activities the fast-growing exotics *Moringa oleifera* L., *Prosopis juliflora* (Sw.) DC and neem (*Azadirachta indica* L.) were planted to improve soil conditions before introducing other native species, since they are well-adapted to saline soils and increase the soil organic-matter content through the production of leaf litter. Native species planted included the medium-growth species soplillo (*Lysiloma latisiliquum*), the slow-growth species *Guaiacum officinale* L. and several other smaller species such as *Colubrina arborescens* (Mill.) Sarg. and sea grape (*Coccoloba uvifera* L.). Exotic fruit species, including peach (*Prunus persica* (L.) Batsch), mango (*Mangifera indica* L.), and coconut (*Cocos nucifera* L.), were also included to provide food for local farmers. All species were incorporated gradually to create a vegetation structure similar to the original vegetation but that also met the food requirements of local communities.

TABLE 13.1.
Characteristics and survival of indigenous species 36 months after planting on the research plots on the farms

Species	Common name in Cuba	Family	Growth characteristics	Functions at site	Survival assessment	
					Mean height (m)	Survival rate (%)
Guaiacum officinale	Guayacan negro	Zygophyllaceae	Slow-growing stabilizing species	Erosion control, animal shelter, wood†	1.23	83
Swietenia mahagoni	Caoba antillana	Meliaceae	Stabilizing species*	Erosion control, animal shelter, wood*	3.03	60
Lysiloma latisiliquum	Soplillo	Fabaceae/ Leguminosae	Colonizing species*	Soil protection†	2.50	57
Conocarpus erectus	Yana	Combretaceae	Slow-growth species*	Soil improvement†	2.40	70
Cordia alba	Uvita	Boraginaceae	Colonizing species*	Soil improvement and protection†	–‡	–‡
Coccoloba uvifera	Uvacaleta	Polygonaceae	Stabilizing species*	Erosion control, fodder†	1.90	17

*Perez and Velázquez (2008).
† Survey findings, 2010.
‡ Not evaluated because the species is a soil creeper.

Results and discussion

Thirteen species (seven native species, three fast-growing exotic species and three exotic fruit trees) were planted on the farms in May and June 2008, taking advantage of existing soil humidity.

Figure 13.2.
The same area shown in Figure 13.1 in 2014, five years after planting

Two kilograms of organic matter made on farm were deposited in each planting hole and mixed with soil. The species were planted as stakes averaging 35–45 cm in height.

After planting, various agro-ecological techniques were applied, including weeding and mulching the soil with weeding residue to reduce evaporation, maintain soil moisture and increase the organic matter around the tree stalks. These measures were taken to promote the development of the root system of plants and their uptake of nutrients.

Survival of planted trees was assessed 36 months after planting. All indigenous species showed a moderate growth after 36 months (Table 13.1; Figure 13.2). These species are characterized by slow to medium growth rates (Bisse, 1988), particularly under the extremely saline soil conditions of the restoration site. Under these conditions it is of great importance to incorporate ground cover species to protect the soil from insolation and promote maintenance of soil moisture. *Swietenia mahagoni* showed the greatest growth

(3.03 m on average) while *Colubrina ferruginosa* had the highest survival rate (92 percent of all planted individuals) (Table 13.1). *Conocarpus erectus* performed well in terms of both survival rate and growth. It also acts as a nursery plant for soplillo (*Lysiloma latisiliquum*), which in turn creates favourable conditions in the lower stratum for the establishment of other species. The high survival rates of soplillo (*L. latisiliquum*), yana (*C. erectus*) and guayacan (*G. officinale*), all native species in the area, indicates that even under severe heat and water stress, certain xerophytic plant species are still able to survive.

Results obtained so far show that it is important to carefully select the species that match the environmental characteristics of the degraded site. Appropriate soil preparation is essential to ensure high survival rates, including very deep plantation holes, application of organic matter and irrigation during establishment. In the 1980s specialists of the Cuban Agroforestry Research Institute established comparable experiments on saline soils with some species used in this study, but without application of organic matter. Rapidly declining survival rates were observed over time (Figure 13.3), which demonstrates the severe environmental pressure plants are under in all these extreme conditions.

Conclusions

The incorporation of a range of native and exotic species increased the diversity of species on the areas to be restored, and created conditions that facilitate the establishment of other species that meet the economic and social needs of local communities. Three years after initiating the study, the results are still preliminary, but show the usefulness of analogue forestry techniques for achieving gradual reforestation and restoration.

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Figure 13.3.
Survival rate (percent) of *Lysiloma havanensis*, *Guaiacum officinale* and *Coccoloba uvifera* in saline soils without organic matter added, obtained from experiments conducted in the 1980s

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13.2. Post-establishment enrichment of restoration plots with timber and non-timber species

David Lamb

Centre for Mined Land Rehabilitation, University of Queensland, Australia

The term "enrichment planting" is used here to describe the situation in which the species in an existing natural forest or plantation are supplemented by adding additional species. The most common reason for undertaking enrichment planting is to increase the proportion of trees that have a commercial value. The technique has been used in logged-over natural forests where natural regeneration has been insufficient and the purpose has been to increase the density of commercially attractive timber species (or, where the species is already present, to increase the density of these commercially attractive trees). An outline of the silvicultural issues is given by Baur (1964), Lamb (1969), Appanah and Weinland (1993), and Dawkins and Philip (1998). Enrichment planting has also been carried out in natural forests in the tropics to create what some have called agroforests (Michon, 2005). In these cases enrichment with food, medicinal or resin-producing species has been done during the fallow stage of shifting cultivation to improve the supply of these products to the forest owner. Such forests are found in many parts of the tropical world and often cover large areas (Clarke and Thaman, 1993; Michon, 2005). Finally, enrichment planting has been used to modify the composition of some planted forests although this has probably not been as widely implemented as in the case of natural forests.

Enriching planted forests

There are two situations in which the enrichment of planted forests can be attractive. One is where the objective is to increase the commercial value of the forest, while the other is where the intent is to increase the conservation value of the forest.

The commercial value can be enhanced in several ways. The first is where a species producing a non-timber forest product (NTFP) can be grown in the shade of existing plantation trees and produce a commercial crop in a shorter time than it would take for the trees to reach a harvestable size. This can make tree-growing attractive to landholders because of the earlier-than-normal cash flow. Examples of this are the growing of rattans, medicinal plants or food crops in plantation understories (Lamb, 2011).

A second form of commercial enrichment is when a plantation monoculture is enriched with timber trees rather than NTFP species. The approach has been used when the preferred timber species cannot tolerate the present site conditions and the initial plantation trees are used to modify these conditions to facilitate the establishment of the preferred species. This situation can arise when native species are being established at degraded sites. This process might be thought of as being less a case of enrichment and more a case of using the initial plantation as a

TABLE 13.2.
The types of species that might be used in enrichment and the types of forests to which these might be added

Types of species used to enrich an existing forest	Types of forest that could be enriched
NTFP species	Monoculture or mixed-species timber plantation
Timber trees or NTFP species	Monoculture plantation of nurse trees
Native species with high conservation value	Monoculture plantation
Timber trees or NTFP species	Disturbed (logging, shifting cultivation, etc.) natural forest
Native species with high conservation value	Disturbed (logging, shifting cultivation, etc.) natural forest

NTFP = non-timber forest product

temporary nurse crop. However, the silvicultural issues confronting the manager are similar. An example of this approach is where the nurse crop or facilitator species is established to shade out weeds, improve the soil fertility or provide some early shade for the target species (e.g. McNamara *et al.*, 2006). In other cases the nurse crop is used to create an environment that reduces insect damage to the target species (Keenan, Lamb and Sexton, 1995). In all of these situations the nurse trees in the original plantation are eventually removed leaving the now-established commercially attractive species to grow.

Enrichment of a planted forest to increase its conservation value is beginning to be practised in situations in which changes in environmental attitudes (or in timber markets) mean that some timber plantations are being taken out of the production forest estate and being added to the protection or conservation forest area (Knoke *et al.*, 2008; Lamb, 2011). Timber plantations close to urban areas or in locations that are strategically important for biodiversity conservation (e.g. adjacent to national parks) may be destined for enrichment of this type. A summary of some of the situations in which enrichment might be practised is shown in Table 13.2.

Methods for undertaking enrichment in natural forests

Some of the principles for undertaking enrichment have been developed from silvicultural research conducted when enriching natural forests. In all cases the key task is to create conditions on the forest floor in which the introduced seedlings can establish and grow (or, where species are added as seed, can germinate and grow). This usually means manipulating the canopy to improve the light environment. Those enriching disturbed natural forests have found that the best time to undertake enrichment is immediately after logging (or, in the case of shifting cultivation, after the cropping period is mostly complete) when the canopy openings are greatest. But even under these ideal conditions it is usually necessary to remove competing vegetation around the planted seedlings at least several times a year for one or two years to allow the planted seedlings to thrive. If circumstances dictate that the process of enrichment is delayed then it is best carried out by cutting strips through the forest and planting seedlings along these strips. Alternatively, it can be done by poisoning or ring-barking a group of trees to create large canopy gaps. In both cases the objective is to manipulate the forest canopy to allow sufficient light to reach to forest floor. Some prescriptions are outlined in Box 13.1.

Some experimentation is usually needed. Large canopy gaps may help newly planted seedlings become established but they will also encourage the growth of competing weed species. A balance has to be struck between opening the canopy enough to promote the growth of the introduced seedlings but not so much that the costs of weed control becomes excessive.

Methods for undertaking enrichment in planted forests

Similar constraints apply when enriching planted forests. However, the methods used depend on whether the objective is to add diversity to a plantation or to increase the commercial value of the plantation. Where the purpose is to add diversity the emphasis is likely to be more on ensuring seedling survival than on maximizing seedling growth. This means that the primary task is likely to be one of creating conditions allowing the newly planted seedlings to simply survive rather than to manipulate the canopy cover to enhance their subsequent growth. The most common approach is to create canopy gaps and plant seedlings of the desired species in these gaps. Just how large the canopy openings must be will depend on the height of the canopy trees, the solar elevation (i.e. on latitude) and on the shade tolerance of the seedlings being planted. Dawkins and Philip (1998) note that wildlife herbivores can sometimes damage newly planted seedlings in existing tropical forests, and Bergquist, Lof and Orlander (2009) and Beguin, Pothier and Prévost (2009) note similar results in forests in Sweden and Canada, respectively. Control of herbivores may be a necessary management activity in many enrichment programmes. This has been done using fences although this method can be expensive.

Not all enrichment need be carried out by planting seedlings and there may be scope for directly sowing seeds on the forest floor of a plantation. Direct sowing has the significant advantage that it can be cheaper to undertake than raising seedlings in a nursery and planting these in the forest. On the other hand, simply broadcasting seeds on the floor of a plantation can be an inefficient use of seeds, since many trials have shown the success rates with direct seeding can be low (Hau, 1997; Engel and Parrotta, 2001; Doust, Erskine and Lamb, 2008). This means it may only be a suitable technique to use with species that are easily available and cheap to collect.

When the species in planted forests are being supplemented for commercial reasons there is usually more emphasis given to maximizing growth as well as survival rates. Again, gaps can

Box 13.1.
Principles for enrichment planting in disturbed natural forest

- The species used must be capable of fast growth, meaning that most will be light-demanding.
- Seedlings should have well-established roots, meaning that container-grown seedlings are preferable to bare-rooted seedlings or wildlings.
- Species should have a low crown ratio (ratio of crown diameter to stem diameter).
- These species should be self-pruning and have good form.
- Planting lines should be oriented in an east–west direction and be separated by a distance about the same as the crown diameter of the species when mature (e.g. around 10–15 m).
- Seedlings should be planted more closely along these lines (i.e. <10 m) to allow for deaths and perhaps thinning.
- All overstorey competition should be removed before planting to avoid damaging young seedlings.
- Weeds along the planting line should be removed at least three times in the first year in a strip about 2 m wide.
- The technique will fail if seedlings are susceptible to grazing by wildlife.
- The regrowth between the planting lines should not be flammable.

Source: Based on Lamb (1969), Appanah and Weinland (1993), and Dawkins and Philip (1998).

be created to increase the amount of light on the forest floor. These gaps must be sufficient to allow the growth of the planted seedlings but not so great as to encourage weed growth. In many cases gaps are created by removing every second or third row of planted trees. The underplanted seedlings used to enrich the plantation are then planted along this line.

Care is needed to ensure that this degree of canopy cover remains appropriate. For example, seedlings may benefit from a certain amount of shade when they are young but require rather more light once they are established. In the absence of extra light their growth may decline and stagnate. This means there can be a trade off between the beneficial advantages of the cover crop, such as providing early shade when the seedlings are sensitive to full sunlight, and then hastening shoot growth rates once seedlings have passed beyond this stage (Keenan, Lamb and Sexton, 1995; McNamara *et al.*, 2006).

Case study: enrichment of monocultures in western Europe

In recent years there has been increased interest in adding additional species, mostly broadleaves, to the simple monoculture forests found in many parts of western Europe (Hansen and Spiecker, 2005; Harmer, Thompson and Humphrey, 2005). Many of these monocultures involve exotic confers such as Sitka or Norway spruce. Silviculturalists have been motivated to add additional species because enrichment can improve biodiversity conservation and also increase resilience, which means the stands are better protected against biotic and abiotic changes. Enrichment can be carried out by planting seedlings of the desired species or by allowing natural regeneration of these species to occur. In both cases better survival and growth is dependent on improving the light environment for the new seedlings. The way the light environment is manipulated depends on the nature of the existing stand but involves creating gaps in the existing forest canopy or removing rows of trees. In some cases these have been commercial fellings, but in others the trees have been felled to waste. Clear felling is usually avoided to prevent sites from being over-run with weeds. Herbivores have been excluded by fencing or by culling. Targets for conversion are usually unstated but may involve reducing the existing dominants to less than 20 percent of the tree density.

Case study: enrichment of logged-over natural forest in Sabah, Malaysia

Large areas of natural forest in Sabah have been degraded by intensive logging and fire. The intensity of these disturbances limited the capacity of the forests to recover naturally. This means it will be some years before another timber harvest will be possible. Enrichment has been carried out to accelerate this recovery process (Yap, 2011). Ideally this should have taken place immediately after the disturbances but the extent of the area needing treatment meant that many areas could not be treated. Instead, these have been occupied by natural regeneration of mainly pioneer species, especially *Macaranga* sp. Enrichment involved girdling or ring-barking these trees and planting seedlings of more than 100 commercially valuable native timber tree species in rows or in gaps, depending on the presence and distribution of native timber trees. Where seedlings are planted in rows, the rows have been about 10 m apart with the seedlings spaced 3–5 m apart along the rows. When seedlings are planted in gaps the aim is to have a cluster of three seedlings in a 10 × 10 m plot. Some tending is carried out to enable seedlings to become established and the planting areas are surveyed after three months to ensure survival is greater than 90 percent (replanting is done if survival is less than this). In the first two years around 2–3 tendings are usually done with less thereafter. Tending can continue for up to ten years depending on need. Across Sabah about 45 000 ha have been enriched using similar methods.

Monitoring

What constitutes success? Anyone undertaking enrichment should have a clear idea of what is an acceptable outcome and when they should intervene to ensure enrichment is succeeding. The most obvious measure of success is that a high proportion of the planted seedlings have survived. But equally important might be that these seedlings are uniformly distributed across the forest area and not clustered in a single small area. Rapid seedling growth may – or may not – be an important additional factor. When enrichment is for commercial purposes the growth rates of the newly planted trees will have to be monitored so that the canopy cover can be adjusted when necessary. But for enrichment planting programmes aimed at improving forest biodiversity, perhaps the most important indicator of success is that the species used to enrich the site have been able to flower, fruit and regenerate themselves on the forest floor. Once this occurs, a simple even-aged plantation begins to be transformed, via enrichment, into a self-sustaining uneven-aged forest. Any monitoring programme needs to be designed as a series of questions such that the answers will give unequivocal guidance to the manager (e.g. Have at least 80 percent seedlings survived? Are these seedlings uniformly distributed across the site? Have the species used in the enrichment programme flowered and produced seed?).

Conclusion

Enrichment planting can be used to add species to existing natural forests and plantations. Much of what we know about the silvicultural techniques needed to undertake enrichment comes from work carried out in degraded natural forests. The primary task is to create canopy openings sufficient to allow light to reach the forest floor and promote seedling growth. When enrichment is being carried out to improve the conservation value of the forest the amount of light needed is that sufficient to allow the seedlings to survive. But when enrichment is being carried out to generate a commercial benefit it may be necessary to continue to monitor and manipulate this light environment to ensure rapid seedling growth.

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13.3. Enrichment planting using native species (*Dipterocarpaceae*) with local farmers in rubber smallholdings in Sumatra, Indonesia

Hesti L. Tata,[1,2] Ratna Akiefnawati[2] and Meine van Noordwijk[2]

[1] Forest Research and Development Agency, Indonesia
[2] World Agroforestry Centre (ICRAF), South East Asia Regional Office, Indonesia

Indonesia has the world's third largest area of tropical forest. An estimated 50 percent of the country's total land area still has forest cover (FAO, 2005). Natural forests in the lowland of Sumatra and Borneo are dominated by *Dipterocarpaceae*, which is one of the most important families for good-quality timber. Some species provide non-timber forest products, such as dammar resin, camphor and illepe nuts. The family consists of 16 genera and is widely distributed from Africa (Congo, Côte d'Ivoire, Ghana, Guinea, Madagascar and the Seychelles), to Asia (the Andaman Islands, India, Indonesia, Malaysia, Nepal, Pakistan, Sri Lanka, Papua New Guinea and the Philippines) and South America (Colombia, Ecuador and Venezuela) (Ashton, 1982; Maury-Lechon and Curtet, 2005). In Sumatra alone, 106 species of *Dipterocarpaceae* have been recorded. Construction timbers derived from *Dipterocarpaceae* include red meranti, white meranti, yellow meranti and bangkirai (Ashton, 1982).

The nature of forest in Indonesia is rapidly changing, even if cover is being maintained. Indonesia has become the global leader in carbon-dioxide emissions from land-use change as a result of the rapid loss of forest biomass and destruction of peatlands (Archard *et al.*, 2002; de Fries *et al.*, 2002). The overall loss of forest cover in Indonesia from 2003 to 2006 was 1.2 million ha/year (MoFor, 2010). In the Bungo district of Jambi province alone, forest cover decreased by 9964 ha/year in the period of 1988–1993, but by only 1211 ha/year in the period of 2002–2005. Between 1988 and 2005, almost 40 percent of the Bungo area was converted to intensive agriculture, such as rubber and oil palm plantations. Rubber trees are planted in both monoculture and agroforestry systems. Between 1973 and 2005, the area under rubber agroforest in Bungo decreased from 15 to 11 percent, while the area under monoculture plantations increased from 3 percent to over 40 percent (Ekadinata and Vincent, 2011).

Although rubber monoculture systems using clonally propagated rubber trees can produce large amounts of latex, agroforests can provide multiple environmental services while ensuring farmer livelihoods (Tomich *et al.*, 2002; Schroth *et al.*, 2004). Rubber agroforest consists of mixtures of rubber trees with other species that regenerate naturally from seed banks or dispersal agents. Some important species, such as fruit trees, are deliberately planted (Joshi *et al.*, 2002). Rubber agroforests range in intensity from secondary forests with some rubber (e.g. 5–10 percent of tree basal area) to vegetation dominated by rubber with a complement of native forest trees. So-called "complex agroforest" systems are characterized by a substantial (but less than 50 percent) proportion of rubber trees in the total biomass and a high diversity of native forest trees and understorey plants (Gouyon, de Foresta and Levang, 1993).

To counterbalance the high rate of deforestation, the Government of Indonesia has initiated tree-planting efforts during the last three decades. Tree plantings using exotic and fast-growing species, such as brown salwood (*Acacia mangium* Willd.) and *Eucalyptus* spp., would provide resources for pulp and paper industries. Some forest rehabilitation is based on enrichment planting with native tree species, such as *Dipterocarpaceae* species (Nawir, Murniati and Rumboko, 2007). Dipterocarp seedlings tend to be shade-tolerant, so are suitable to be planted in an agroforestry system with rubber trees. Planting dipterocarp trees helps to meet the challenge posed by domestic demand for timber, despite being constrained by rules and regulations on extracting hardwood from farm-forests.

Several studies have been conducted on enrichment planting in rubber plantations with *Dipterocarpaceae* in various areas of Bungo and Tebo districts, Jambi province (Anonymous, 2004; Tata *et al.,* 2010). These have shown that dipterocarp species grow well in rubber plantings and do not suffer from mycobionts and abiotic factors such as soil and microclimate (particularly light availability).

Here we report on the early growth of meranti in rubber agroforests in three villages in Bungo district and farmers' participation in tree enrichment planting in rubber smallholdings.

Activities of enrichment planting

Study site

Bungo district is located in western Jambi Province, Sumatra, Indonesia. Bungo has the third largest area of rubber agroforest in Indonesia. The sites were selected based on degree of land intensification: (i) low intensification (with forest and complex rubber agroforests dominating the landscape) was represented by Lubuk Beringin village; (ii) intermediate intensification (with complex to simple rubber agroforests dominating) was represented by Tebing Tinggi village; and (iii) high intensification (with simple rubber agroforests, monoculture rubber and oil palm) was represented by Danau village (Therville, Feintrenie and Levang, 2011) (Figure 13.4).

Farmer selection

One farmer was selected at each site based on willingness to collaborate. The farmers are aware that wood stocks are getting scare, owing to few remnant natural forests and lack of wood supplies from plantations to meet local demand. Staff from the World Agroforestry Centre (ICRAF) provided technical support to farmers to familiarize them with the different character and silvicultural requirements of dipterocarp species compared with rubber trees. An earlier study showed that very few rubber farmers (about 12.5 percent of respondents) had experience in planting forest-tree species (Tata and van Noordwijk, 2012). Farmers were responsible for regular seedling maintenance in the plots.

Rubber agroforest development and maintenance

Establishment of rubber agroforests begins with slashing the forest cover and burning it during the dry season. This method is relatively cheap and commonly applied by farmers in the area, in part because they believe that ash improves soil fertility (Ketterings *et al.*, 1999). The plots in the three sites were established 10–12 years ago, at a planting distance of 6 × 3 m or 6 × 5 m. Rubber trees were being tapped by the time *Shorea leprosula* seedlings were being interplanted in the rubber plots.

Figure 13.4.
Sites where dipterocarp species were planted in rubber agroforests in Bungo district, Jambi

Tree species and planting

Shorea leprosula is a native species that grows in the lowland forest of Sumatra. It is known locally as *meranti batu*. *S. leprosula* wood is sold as red meranti, and is used for light construction, furniture and moulding. The species can be grown in various soil types, from fertile to poor. It has a long life cycle; wood can be harvested 20–25 years after planting. *Shorea leprosula* grown on Ultisol soil in central Kalimantan grows by about 3.2 cm/year, and hence is classified as a fast-growing meranti (Soekotjo, 2009).

Seedlings were bought from an uncertified vendor in Sungai Duren village, Jambi. Wildings were collected from the surrounding remnant forest areas. Therefore the age and the origin of mother trees of the wildings were unknown. The seedlings were planted between rubber trees in the rubber gardens at a spacing of 10 × 7 m. The number of *S. leprosula* seedlings planted depended on the area of the rubber garden, ranging from 48 to 70 trees per plot. All farmers actively maintained the *S. leprosula* seedlings, weeding an area around them, but applied no fertilizer.

Dipterocarps form symbioses with ectomycorrhizal fungi, but meranti do not need to be inoculated with the fungi to establish in tropical forest (Lee, 2006; Tata *et al.,* 2010). This proved to be the case in this study; *S. leprosula* seedlings were not manually inoculated with ectomycorrhizal fungi but most of the roots of seedlings were naturally inoculated by unidentified ectomycorrhizal fungi.

Monitoring and experiences

Survival of* S. leprosula *in rubber plots

Survival rate of *S. leprosula* six months after planting ranged from 46.5 percent to 59.2 percent in the three plots and remained the same at 12 months after planting (Figure 13.5). Survival rate was lowest at the Tebing Tinggi site because wild pig (*Sus scrofa*) attacked both rubber trees and *S. leprosula* trees in the plots. Similar attacks on *Shorea* seedlings in other plots in Bungo and Tebo district were also reported by Tata *et al.* (2010).

Figure 13.5.
Survival rate of *S. leprosula* in three sites in Bungo District, Indonesia

Figure 13.6.
Early growth of *S. leprosula* in three different plots in Bungo districts: Tebing Tinggi, Lubuk Beringin and Dusun Danau: (a) diameter growth, (b) height growth

Note: vertical line shows standard error.

Early growth of S. leprosula *in rubber plots*

Height growth was greatest in Dusan Danau while growth in stem diameter was greatest in Dusan Danau and Tebing Tinggi (Figure 13.6). The poor growth of *S. leprosula* in Lubuk Beringin was the result of poor maintenance, particularly lack of weeding, by the farmer at that site.

Farmer participation

Many rubber farmers are reluctant to plant forest-trees in rubber plots because they believe that rubber and timber trees compete for soil nutrients and light, which reduces production of latex. The farmers who took part in the current study were willing to do so because: (i) they received free seedlings and technical assistance, (ii) they were aware of the shortage of timber in their areas, (iii) they could use the planted trees as a means of saving and as collateral for credit, (iv) they were innovators and (v) because they received recognition from others (Tata and van Noordwijk, 2012).

The participating farmers also planted other trees, including *Litsea* sp., bitter bean (*Parkia speciosa* Hassk.) and *Archidendron jiringa* (Jack) I. C. Nielsen. *Litsea* sp., which produce light timber and usually regenerate naturally, while *P. speciosa* and *A. jiringa*, which are grown for their fruit, are usually planted but can regenerate naturally in rubber plots during the fallow period. Although *S. leprosula* is a native species and produces good timber, it is not commonly planted by the farmers in rubber plots. Farmers are mostly interested in planting exotic species, such as teak (*Tectona grandis* L.f.) and big leaf mahogany (*Swietenia macrophylla* King) because of their market value. Market access is one of the key reasons why a farmer will plant a commodity tree. Farm forests of teak, albizia *(Paraserianthes falcataria* (L.) Nielsen) and some other timber species are already well established and supported by the Government of Indonesia. In contrast, dipterocarps such as *Shorea* spp. can be harvested for sawn wood only in forest-concession areas, that is, industrial plantation forest (*hutan tanaman industri*) and community plantation forest (*hutan tanaman rakyat*), and not from farm forests (*hutan rakyat*). This restriction on harvesting, transporting and marketing timber of dipterocarps from farm forests, such as rubber agroforest, hampers forest restoration using native species.

Conclusions

Dipterocarp species native to Indonesia are recommended for use in ecosystem restoration in Sumatra. Red meranti (*S. leprosula*) grows well in the rubber agroforestry systems in Bungo district, Jambi province. However, active participation of farmers in restoration activities is essential to achieve high survival rate and performance of the planted seedlings. Changes to government regulations are required to permit harvesting, transport and marketing of red meranti from farm forests, such as rubber agroforests.

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13.4. "Rainforestation": a paradigm shift in forest restoration

Paciencia P. Milan

Visayas State University, Philippines

With less than 24 percent of its forest cover remaining, the Philippines is experiencing loss of ecosystem services such as biodiversity maintenance, carbon sequestration, watershed protection and forest products for local communities. As a result, natural disasters such as flash floods, landslides and even water shortages have increased. Despite measures taken to curtail forest destruction, the forests continue to decline. Most reforestation efforts in the Philippines focus on the development of forestry and agroforestry systems using tree species selected for their fast growth and easy germination. Species composition of the forest that covered the land prior to logging is rarely taken into account (Margraf and Milan, 1996).

The use of non-native species in forest restoration impacts greatly on forest biodiversity in the Philippines. Native tree species are being lost from the landscape because their timber is still sought, especially for construction. They continue to be cut even if other types of timber are available. However, in spite of their popularity for their high wood quality, native or local trees are not propagated or used in reforestation. Hence, mother trees have become rare, which in turn reduces the availability of propagation material. As the rainforest tree species are depleted and monoculture of exotic or introduced species in reforestation proliferates, the survival of local wildlife species is at stake. Some of them play an important role in pollination and seed dispersal (Hutter, Goltenboth and Hanssler, 2003). The repeated clearcutting of fast-growing exotics on reforestation sites rapidly exhausts soil nutrients and, to some extent, water, making reforestation increasingly difficult. Another drawback of monoculture of exotic species such as *Gmelina*, mahogany and *Leucaena* is their vulnerability to pests.

A paradigm shift in reforestation is needed to achieve sustainability. As reforestation in the Philippines can be described as a failure in terms of restoring original vegetation, an innovative strategy known as "rainforestation" has been developed through a joint research project with the Philippine-German Tropical Ecology Program, a bilateral project between the German Organization for Technical Cooperation (GTZ, now the Deutsche Gesellschaft für Internationale Zusammenarbeit, GIZ) and the Visayas State College of Agriculture, the Philippines (now Visayas State University) in 1996.

TABLE 13.3.
Shade-loving local forest tree species of Leyte, Philippines, recommended for rainforestation on volcanic soils

Local Name	Scientific Name
Palosapis	*Anisoptera thurifera* (Blanco) Blume
Apitong	*Dipterocarpus grandiflorus* Slooten
Hairy Apitong	*Dipterocarpus philippinensis* Foxw.
Hagakhak	*Dipterocarpus warburgii* Brandis
Dalingdingan	*Hopea foxworthyi* Elmer
Yakal-kaliot	*Hopea malibato* Foxw.
Almon	*Shorea almon* Foxw.
Guijo	*Shorea guiso* Blume
Yakal-malibato	*Shorea malibato* Foxw.
Red lauan	*Shorea negrosensis* Foxw.
Tangile	*Shorea polysperma* Merr.
Kamagong	*Diospyros philippensis* (Desr.) Gürke

The rainforestation concept

Only indigenous and native forest tree species are used in rainforestation. This approach emphasizes improvement of the structural habitat to support wildlife. It consists of three operational frames: habitat restoration, biodiversity conservation and provision of ecological services. Rainforestation is more consistent with biodiversity conservation strategies such as protected-area management and natural regeneration than conventional reforestation efforts.

The planting scheme in the restoration areas involves planting sun-loving native tree species at a close spacing of 2 × 2 m to shade out weeds. Species used on limestone soils include *Terminalia microcarpa* Decne. (known locally as *kalumpit*), *Calophyllum inophyllum* L. (*bitaog*), *Vitex parviflora* Juss. (*molave*), *Melia dubia* Cav. (*bagalunga*), *Dracontomelon dao* (Blanco) Merr. & Rolfe (*dao*), *Calophyllum blancoi* Planch. & Triana (*bitanghol*), *Vitex pinnata* L. (*lingo-lingo*) and other fast-growing pioneer species. Shade-loving trees (Table 13.3) are planted between rows when the pioneer trees start to provide shade. Selection of pioneer tree species depends on the soil type, whether limestone or volcanic.

Once these tree species have established, some specialized arthropods, birds and lizards stage a comeback. Over time, biodiversity increases with the number of native tree species and the structural diversity it offers to wildlife (Milan, 1996). In addition, the closed forest system promotes nutrient cycling. Aside from biodiversity improvement of rainforestation farms, soil properties, biological activities and microclimate improve noticeably. Soil pH and structure improve, increasing water-holding capacity. In calcareous soil (Punta) and the demonstration site of the Visayas State University, soil colour changed from light sandy to dark brown or black, and soil organic matter and nutrient content increased, especially nitrogen, calcium and magnesium. Soil microclimate improved becoming moist and cooler, and soil arthropods and other fungi proliferated (Asio *et al.*, 1995).

Figure 13.7.
Example of the combination of native tree species and fruit trees in rainforestation farming

Source: adapted from Margraf and Milan (1996).

Over almost two decades rainforestation believers have been raising planting materials of native tree species in nurseries across the Philippines. Most of those trained in nursery management took up growing native tree seedlings as a livelihood endeavour.

In spite of advocacy for the use of indigenous forest tree species, the use of native tree species in reforestation programmes has received very little support, mainly because of the following perceived limitations:

- Native species, especially dipterocarps, grow slowly.
- Dipterocarps fruit approximately only every three to five years, depending on species and locality.
- Too few seedlings can be produced in a short time because nursery management of native species is not well understood.
- Dipterocarp seedlings and other native species require shade and cannot be used to reforest open areas.

Rainforestation farming

Rainforestation farming is a systems perspective in forest restoration that emanated from the rainforestation concept. It not only preserves forest biodiversity but simultaneously sustains human food production (Milan and Margraf, 1994). In rainforestation farming, native or indigenous tree species are used in combination with agricultural crops and fruit trees (Figure 13.7).

The rainforestation farming system, when appropriately understood and implemented, can replace the destructive form of *kaingin* or slash-and-burn farming, allowing upland farmers to continuously crop even a small area (minimum of 0.05 ha). Planting fast-growing native trees in the first year and premium tree species in the following years contributes to the conservation of local genetic resources. By incorporating fruit trees and other crops, rainforestation farming can provide farmers with additional income. Thus, rainforestation not only contributes to saving forest ecosystems but also helps to address the needs of farmers for food, timber and other forest products in a sustainable way (Goltenboth and Hutter, 2004). As a result, it is acceptable to resource-poor farmers and landowners alike.

Most assessments of the benefits of reforestation focus on the easily measurable economic benefits and ignore non-monetary benefits, such as ecosystem services, which are harder to quantify. Promoting adoption of rainforestation rather than use of traditional approaches to reforestation will depend on achieving a deeper understanding of the interplay between the potential to improve farmer income and the ecological function of the forest biodiversity.

Rainforestation offers the prospect of sustainable development in the uplands and forest ecosystems of the Philippines. This was recognized by the Philippine Government in 2011, when rainforestation was adopted as a strategy in the National Greening Program implemented by the Department of Environment and Natural Resources (Executive order no. 26, 24 February 2011).

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13.5. The permanent polycyclic plantations: narrowing the gap between tree farming and forest

Enrico Buresti Lattes,[1] Paolo Mori[2] and Serena Ravagni[3]

[1] Association for Tree Farming for Economy and the Environment, Arezzo, Italy
[2] Compagnia delle Foreste, Arezzo, Italy
[3] Agricultural Research Council (CRA), Forestry Research Center, Arezzo, Italy

Many of the environmental benefits provided by tree plantations (tree farming) are lost at the end of a management cycle when the trees are felled. Permanent polycyclic[24] plantations, which combine the advantages of plantations with some of those of the forest, can help avoid this disadvantage (Buresti Lattes and Mori, 2009).

Many living organisms, including animals, insects and plants, are associated with each tree. Thus, diversity increases and becomes more complex as the number of trees and tree species and the complexity of the vertical or horizontal structure of the forest increase. This aspect is very important not only in forested landscapes, but also in areas such as intensively cultivated agricultural lands and peri-urban areas, where the presence of trees and shrubs can increase biological diversity. Trees also influence microclimate, regulate water flows and reduce the effect of some pollutants. Moreover, trees sequester CO_2 from the atmosphere. On permanently forested sites, carbon steadily accumulates in the soil and the amount of carbon sequestered in the soil may exceed the amount stored in the plant biomass (Petrella and Piazzi, 2006).

Thus, trees are important not only for productive purposes but also because of their ecological and landscape impacts. Therefore, tree farming, especially polycyclic plantations, has an important environmental role. Polycyclic plantation tree farming covers a wide range of planning and management approaches, from the Italian classical poplar cultivation (AA.VV., 1987) with large farmer inputs and strong impact on the environment to silvicultural approaches with low farmer inputs and little environmental impact. However, all tree plantations generally have the same end point: when the main trees reach the end of their economic life, all trees in the plantation are felled and the ecological and landscape benefits of the plantation are lost.

Recently, researchers have started testing new permanent polycyclic plantations in order to extend the ecological benefits derived from plantations while maintaining profits to the farmer.

What are permanent polycyclic plantations?

Polycyclic plantations are defined as plantations, generally mixed, where there are several groups of main trees with different objectives and lengths of productive cycles. Thus, for example, a classical cloned poplar plantation is monocyclic while a mixed plantation of poplar clones and walnut (*Juglans regia* L.) is a polycyclic plantation.

Recently, it was considered necessary to distinguish polycyclic plantations from permanent polycyclic plantations (Buresti Lattes and Mori, 2006, 2007a). In polycyclic plantations the species with the longest production cycle are planted at a density that allows them to develop a closed canopy at the end of their production cycle; these trees are generally clear cut once they reach maturity. Permanent polycyclic plantations differ from non-

[24] In this chapter polycyclic means the contemporary presence of two or more wood production cycles of different lengths on one plot of land.

permanent plantations in terms of tree spacing and management strategy. Distances between the main trees with a longer production cycle are greater than in polycyclic plantations to prevent development of a closed canopy (Buresti Lattes and Mori, 2007b). This allows trees of the same or different species to be planted between these main trees on a different production cycle, and the land remains continuously under tree cover.

As an example we can consider a mixed polycyclic plantation with poplar clones (but also native species of *Populus alba* L. or *Populus nigra* L.) and walnut. In a polycyclic plantation (e.g. Ravagni and Buresti, 2003; Buresti Lattes and Mori, 2007b, 2009; Buresti Lattes *et al.*, 2008a), when poplar is felled walnut occupies all of the liberated space, and the farmer must wait for the walnut to complete its production cycle before establishing a new cycle with poplar or other (native) species. However, with permanent polycyclic plantations, walnut trees are spaced such that at the end of the production cycle their canopies do not occupy all the available space but instead leave space for subsequent tree populations. Thereby, after poplar felling the farmer may decide to start a new production cycle by planting poplar, or another (native) species according to the environmental conditions and available space. While walnut continues to grow, the farmer may thus be able to produce two or more cycles with other species in the available space. When the walnut trees are finally cut, trees of other species remain in the plantation to buffer the temporary and partial lack of large trees. A new productive cycle can then be started, replacing the harvested trees with walnut or other native species according to the farmer's production aims.

The arrangements and benefits of polycyclic plantations

Permanent polycyclic plantations require careful planning and management that is adapted to the needs of the species and their different production cycles. The planner has to choose the spacing between trees of the same and different production cycles to allow for optimal performance of all trees. The larger the number of production cycles to be combined in the plantation, the greater the complexity of the design.

The farmer has to understand the growth dynamics of the different trees and the timing of each production cycle in order to conduct the management interventions at the appropriate times (e.g. pruning, felling and introduction of the new production cycle). Technical advice is very important during all these operations.

Planning and managing a polycyclic, mixed, multi-objective permanent plantation (PMMP) is certainly more difficult than, for example, planning and managing a normal mixed plantation. However, PMMPs can provide a number of advantages for the farmer (Buresti Lattes, Mori and Ravagni, 2001; Buresti Lattes and Ravagni, 2003; Becquey and Vidal, 2008a, 2008b; Buresti Lattes and Mori, 2010):

- With the right spacing, older trees will influence the form of younger trees, making pruning simpler.
- With overlapping production cycles, income is more frequent and economic return can be higher than from a simpler plantation.
- The plantation can be redesigned after trees have been felled in each production cycle; changes can be made to species, spacing and production objectives and exploitation of the available space improved.

Moreover, permanent plantations provide ecological and other benefits for society that cannot be achieved with traditional plantations and non-permanent polycyclic plantations. These include:

- less change in the landscape over time;
- continuous carbon storage; and
- less habitat change for fauna that depend on trees for refuge and food.

Case study: polycyclic permanent plantations in Mantua, Italy

One of the first experimental PMMPs was established in the province of Mantua, Italy, in 2006. The plantation was established on a farm devoted to poplar cultivation where the owner was interested in producing poplar with fewer external

Box 13.2. Definitions of terms

A **main tree** provides at least one of the main products for which the plantation was designed.
An **accessory tree or shrub** facilitates the management of the plantation by the farmer but can be substituted by cultural care.
Multifunctional tree farming refers to tree cultivation designed to satisfy multiple functions (e.g. timber production and reduction of pollutants into waterways, or, in the case of common walnut, timber and fruit).
Multi-objective tree farming refers to tree cultivation designed to obtain more than one type of wood product (e.g. timber and biomass).

inputs and higher environmental value (Buresti Lattes *et al.*, 2008b).

The planting scheme proposed (Figure 13.8) used main trees of three different tree species: pedunculate oak (*Quercus robur* L.) and poplar (several clones) to produce timber and hornbeam (*Carpinus betulus* L.) to produce biomass for fuel or wood panels. Oak and hornbeam are native to Italy. The objective for the oak was to produce logs of 40–45 cm in diameter and 4 m in length; for poplar the objective was to produce logs of 40 cm in diameter and 6 m in length. Hornbeam was selected not only for the good quality of its wood as biomass, but also for its capacity to grow in partial shade and for its low competitiveness towards oak located only 4.5 m away.

Legend to Figures 13.8 to 13.18

Main plants

Pedunculate oak (*Quercus robur* L.)

Poplar clone

Hornbeam (*Carpinus betulus* L.)

Accessory plants

Black alder (*Alnus glutinosa* (L.) Gaertner)

Buckthorn (*Rhamnus frangula* L.) or viburnum (*Viburnum* spp.)

Figure 13.8.
Year 0. The first plantation scheme of all species.

Figure 13.9.
Year 10. Poplars should have reached the production target of 40 cm trunk diameter and are felled.

Accessory trees and shrubs (all native) were planted along the oak rows to improve shape and reduce branching in young oaks and to mitigate the isolation stress that might occur after harvesting the poplar. The accessory trees and shrubs included black alder (*Alnus glutinosa* (L.) Gaertner), buckthorn (*Rhamnus frangula* L.) and laurustinus viburnum (*Viburnum tinus* L.). Nitrogen fixation by the symbiotic bacteria in black alder roots is also expected to improve the productivity of the oak.

Figures 13.9 to 13.18 describe one possible evolution and management strategy for the plantation over 35 years. Production cycles are expected to be 10 years for poplar, 35 years for oak and 10–15 years for hornbeam. The first cycle of hornbeam is relatively long compared with that of other native species that are suitable for the production of woody biomass. This is because of the relatively slow initial growth of hornbeam and the need to protect the adjacent oak trees. Sudden isolation of the oaks should be avoided as it could cause stress reactions, and therefore surrounding poplar and hornbeam must be felled alternately. After the first ten years, one or other tree species will be harvested every five years. Felled trees are replaced with the same or different species, adjusting the planting distance according to species competition at each stage of development. Over time, location of the species on the site can rotate; where there were valuable timber trees is it possible to plant trees for biomass production and vice versa.

The scheme may change over time, depending on the development of one or more species or changing needs and preferences of the owner.

This plantation scheme is just one possibility. The choice of species and management strategies will depend on environmental conditions, farmer needs and applicable regulations in each area and may change over time.

Figure 13.10.
Year 10. After felling the poplar trees, two new poplar rows are planted. The new poplar rows will be separated by 7 m and will be 10 m from the oaks. The hornbeam trees should not be cut at the same time as the poplars to avoid the excessive and sudden isolation of the oaks, which could cause stress reactions.

Figure 13.11.
Year 15. Hornbeam is felled. The first cycle of hornbeam is relatively long compared with that of other native species that are suitable for the production of woody biomass. This long time is due both to the relatively slow initial growth of the hornbeam and to the need to extend the oak protection for additional years.

Figure 13.12.
Year 20. Poplars should have reached the production target of 40 cm in diameter and are felled.

Figure 13.13.
Year 20. Two new rows of poplar are planted. This time the rows are staggered in order to increase the spacing (trees will be 5 × 5.6 m from each other). The minimum distance from oaks (which are 20 years old and with a well-developed crown) will be 11 m. This distance should be enough to allow poplar to grow without competition with oaks. The two five-year-old hornbeam rows, which are 4.5 m from the oaks, should have a positive effect, protecting oaks from isolation stress.

Figure 13.14.
Year 25. The hornbeam suckers, which grow faster than seedlings, should be ready for felling.

Figure 13.15.
Year 30. The third cycle of poplar should be felled and the hornbeam between the two poplar rows removed.

Figure 13.16.
Year 30. A new row of oaks and accessory trees is planted together with two new rows of hornbeam 4.5 m from the oaks.The two new hornbeam rows are 9 m from the 30-year-old oaks.

Figure 13.17.
Year 35. The oaks should have reached the target size and should be felled. The hornbeam that is closest to the oak trees should also be felled, but yield will be low. At this point, thanks to the positive competition with the older oaks and to the microclimate provided by the oaks, the five-year-old hornbeam plants should be adequately developed.

Figure 13.18.
Year 35. New rows of poplar and hornbeam are planted. Trees for timber production are now planted where trees for biomass production were before, and vice versa.

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Chapter 14

Habitat-specific approaches

14.1. Mangrove forest restoration and the preservation of mangrove biodiversity

Roy R. Lewis III

Lewis Environmental Services Inc., Salt Springs, Florida, United States

Mangrove forest ecosystems covered 13.8 million ha of tropical shorelines in 2000 (Giri *et al.*, 2011), down from 19.8 million ha in 1980 and 15.9 million ha in 1990 (FAO, 2003). These losses represent about 2 percent per year from 1980 to 1990 and 1 percent per year from 1990 to 2000. Therefore, achieving no net loss of mangroves worldwide would require the successful restoration of approximately 150 000 ha per year, unless all major losses of mangroves ceased. Increasing the total area of mangroves worldwide towards their original extent would require an even larger effort.

An example of documented losses of mangroves is the combined losses in Malaysia, the Philippines, Thailand and Viet Nam of 7.4 million ha (Spalding, 1997). These figures emphasize the magnitude of the loss. The opportunities that exist to restore areas back to functional and biodiverse mangrove ecosystems are also significant, including mosquito-control impoundments in Florida (Brockmeyer *et al.*, 1997) (several tens of thousands of hectares) and abandoned shrimp aquaculture ponds in Southeast Asia (Stevenson, Lewis and Burbridge, 1999) (several hundreds of thousands of hectares).

While great potential exists to reverse the loss of mangrove forests worldwide, most attempts to restore mangroves fail completely or fail to achieve the stated goals (Erftemeijer and Lewis, 2000; Lewis, 2000, 2005, 2009). Previously documented attempts to restore mangroves (Field, 1996, 1999), where considered successful, have largely concentrated on creation of plantations of mangroves consisting of just a few species with the objective of providing wood products (Kairo *et al.*, 2002) or collecting eroded soil and raising intertidal areas to usable terrestrial agricultural elevations (Saenger and Siddiqi, 1993).

Restoration of a biodiverse mangrove forest

Successful mangrove forest restoration requires careful analyses of a number of factors before attempting actual restoration. Lewis (2005, 2009) notes that existing hydrology of a proposed restoration site needs to be characterized and compared with that of a reference forest to establish what conditions preclude natural recovery in damaged forests, or what conditions prevent natural recolonization of supratidal and subtidal flats that might be proposed for conversion to mangrove forests. A six-step process called ecological mangrove restoration has evolved from earlier attempts to standardize successful approaches (Stevenson, Lewis and Burbridge, 1999), and is now taught around the world (Lewis 2010). This method emphasizes getting the hydrology right first and then observing and documenting natural recovery through volunteer mangrove propagule recruitment (Figure 14.1) before large-scale planting of mangroves is even considered. As seen in Figure 14.1, mangrove propagules can voluntarily recruit to a restored site and establish the natural biodiversity of mangrove species. Planting is therefore not needed in most cases. Situations where planting of mangroves is needed are described as "propagule limited" sites (see below).

Unfortunately, as noted in Stevenson, Lewis and Burbridge (1999) and Samson and Rollon (2008), massive attempts to plant mudflats where mangroves have never existed have been the norm for many decades and have almost uniformly failed. Where they occasionally do work because local topographic conditions are conducive to planting of mangroves, the results are typically plantations of a single species of mangrove. Various species of *Rhizophora* are commonly used in plantings as they have large propagules that are easily collected, grown and planted. This emphasis on single-species plantings ignores the mix of species found in most mangrove forests. Mangrove forests in the New World typically contain four species of mangrove, and a single forest in a location such as the Philippines, Viet Nam and northern Australia may contain up to 30 species (Duke, 1992). There are 69 species worldwide called mangroves (Duke, 1992).

Biodiversity is also threatened by the introduction of non-native species of mangroves for restoration. Chen *et al.* (2009) notes that *Sonneratia apetala* Buch.-Ham. has been introduced to China from Bangladesh and, surprisingly, used to control another introduced plant species, *Spartina alterniflora* Loisel, "even though the invasiveness of this exotic mangroves species was not fully understood" (Chen *et al.* 2009: 49).

An important goal of many restoration projects is to provide habitats for fish and invertebrates to restore local fisheries. Maximizing use of such habitats usually means maximizing biodiversity of the plant species, and therefore a monotypic stand of mangroves in an area that normally supports 20 or more mangrove species is not a logical goal. Establishment of persistent tidal creeks to assist with entry and exit of juvenile and adult fish and invertebrates is also an essential restoration objective. Lewis and Gilmore (2007) discuss the use by fish of both natural and restored mangrove forests and report specifically on monitoring a successful 500 ha mangrove restoration project in Hollywood, Florida, United States (see Figure 14.1), where fish populations sampled in both reference and restored sites were statistically indistinguishable within three to five years of restoration. They emphasize three restoration and design goals to ensure functional and naturally biodiverse ecological restoration of mangrove forests:

1. Achieve plant cover similar to that in an adjacent relatively undisturbed control area of mangrove forest.
2. Establish a network of channels that mimic the shape and form of a natural tidal creek system.
3. Establish a heterogeneous landscape similar to that exhibited by local mangrove ecosystems.

Lewis (2005) introduced the term "propagule limitation" to define a condition in which natural recovery is slowed or halted because no natural mangrove propagules are available to volunteer at a damaged site. The absence of propagules may be caused by a large-scale loss of adult trees capable of producing propagules or by hydrologic restrictions or blockages (e.g. dykes) that prevent natural waterborne transport of mangrove propagules to a restoration site. Since propagules are produced at different times of the year by different species in different locations (Tomlinson, 1986), more than one site visit may be necessary to correctly identify a propagule limited site. Lack of propagules at a single time of year does not necessarily define a propagule-limited site, and therefore careful evaluation of this parameter is important. If a damaged forest will recover on

Figure 14.1.
Time sequence photographs of a portion of the 500 ha West Lake Park mangrove restoration project utilizing non-native exotic plan removal, site excavation, tidal creek restoration and natural recruitment of mangrove propagules. No planting of mangroves took place.

its own within an acceptable time frame, any attempt to introduce propagules, plant propagules or plant nursery-grown mangroves is likely to be a waste of time and money. Recovery is here defined as the recolonization of a restoration site and growth of plant materials on that site reaching some predefined numerical target (e.g. percent cover, total basal area). Priority should be given to restoration sites that would indeed benefit from human intervention at the least per unit cost, given that time and money to devote to any restoration project are always limited.

These suggestions may seem obvious, but there are very few documented examples of successful mangrove forest restoration. More commonly, well-intentioned, but often faulty, mangrove restoration efforts target areas on which mangroves were not previously present, such as mudflats or seagrass meadows seaward of natural mangroves or damaged areas without a properly documented history (Field, 1996; Erftemeijer and Lewis, 2000; Lewis, 2005). The result of unsound evaluations of restoration opportunities has, unfortunately, emphasized first establishing a mangrove nursery and then planting mangroves at a casually selected site as the primary tool in restoration, rather than first assessing the reasons for the loss of mangroves in an area and working with the natural recovery processes (Lewis, 2009).

Both Brockmeyer *et al.* (1997) and Stevenson, Lewis and Burbridge (1999) present examples of successful mangrove restoration following re-establishment of historical tidal connections to adjacent estuaries. This is termed "hydrologic restoration" (see discussion in Turner and Lewis, 1987). In the examples discussed, volunteer propagules of mangrove and mangrove nurse-plants were sufficient to allow for rapid establishment of plant cover. No planting of mangroves was required.

Establishing success criteria

Once a site is finally chosen for restoration and a design developed, quantifiable success criteria should be established. Establishing such criteria is important in order to actually measure progress towards successful restoration. The first step in establishing numeric criteria for success is to prepare a brief narrative goal or set an objective for the project (Saenger, 2002). This will define the next steps. For example, a goal may be to establish a monotypic plantation of *Rhizophora apiculata* Bl. to be harvested after 12 years as poles. It may be an acceptable goal to local stakeholders in the project, such as local villages and fishermen, and harvest of wood products from locally managed forests is a typical goal (see discussion of timber production in the Matang Forest, Malaysia, in Saenger [2002]: 231–234).

A second example of a goal might be to maximize biodiversity. In this case, the restoration site might be left alone and not planted immediately to allow for volunteer colonization of the largest number of different species of mangroves from propagules produced by trees adjacent to a restoration site.

The next step is to look at available information on both plantation and natural recruitment indices of success. Saenger (2002: 256–270) discusses in great detail what is to be expected in terms of biomass and stem density, for example, from typical plantation projects. There has been much work on plantation projects in which just a few species of mangroves are managed, and thus there is a wealth of data to examine. In contrast with this, data on natural recruitment within a mixed forest are generally not available. McKee and Faulkner (2000) report on the results of sampling for density and basal area within two restored mangrove forests in Florida, United States, and compared these with two adjacent control areas. Their data show that density and basal area of volunteer mangroves in the restoration areas exceeded that of planted mangroves. Proffitt and Devlin (2005) report similar results from one of the same sites sampled by McKee and Faulkner (2000) but that they sampled in later years as the system matured. Lewis, Hodgson and Mauseth (2005) report on the results of cover sampling over a period of five years within a restored mangrove forest in another location in Florida, United States. These studies help define parameters that need to be sampled and sampling methodologies, but provide limited data to apply to local situations in other parts of the world.

Few studies exist on trends in biodiversity in restored mangroves, and the range in age, species and inundation class of restored sites makes generalizations difficult. However, the co-occurrence of many animal species in both restored and comparable natural forests suggest that colonization of restoration sites by both mobile and non-mobile fauna is a rapid process, and equivalent populations of mangrove fauna in both natural controls and restored mangrove sites can typically be found within 5–10 years of restoration (Lewis and Gilmore, 2007; Bosire *et al.*, 2008).

Concluding remarks

Restoration of mangrove forest has not been generally successful except where timber production was the goal and monotypic stands were established. Establishment of a biodiverse mixed-species forest cover and restoration of functions equivalent to those of an adjacent reference forest, have not typically been design criteria, and most restoration projects with some general ecological goals have not been successful (Erftemeijer and Lewis, 2000; Lewis, 2005). The chosen restoration sites for many of these projects have been mudflats or seagrass beds lying seaward of the outer edge of existing mangrove forests. These sites are typically planted with nursery-grown mangrove seedlings which do not survive because of frequent inundation and waterlogging.

Although there are relatively few studies on trends in biodiversity in restored mangroves, it appears that colonization of restoration sites by both mobile and non-mobile fauna is a rapid process that may take 5–10 years to reach levels comparable to natural sites (Bosire *et al.*, 2008). The scientific basis for optimum design of restoration projects to meet certain established criteria, such as increased fish production or more use by wading seabirds, is, however, very minimal.

In future, mangrove restoration projects should be more carefully designed to ensure successful establishment of a biodiverse plant cover over large areas at minimal cost. This can be achieved, for example, by restoring hydrologic connections to impounded mangrove areas, as has been done in Florida (Brockmeyer *et al.*, 1997), Costa Rica and the Philippines (Stevenson, Lewis and Burbridge, 1999) using the basic principles of ecological mangrove restoration (Lewis, 2010). Use of non-native species of mangroves in management and restoration projects should be avoided.

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14.2. Forest restoration in degraded tropical peat swamp forests

Laura L.B. Graham and Susan E. Page

Department of Geography, University of Leicester, United Kingdom

Tropical peat swamp forests and their degradation

Tropical peatlands in Southeast Asia are the most extensive in the world; they contain ~69 Gt of carbon, equivalent to 11–14 percent of global peatland carbon, and cover 247 778 km^2, the majority being in Indonesia (206 950 km^2; 57 Gt of carbon;

Page, Rieley and Banks, 2011). In addition, tropical peat swamp forests provide a range of other environmental services including a habitat for endemic, endangered and rare species (Posa, Wijedasa and Corlett, 2011) and have a role in regional hydrological regulation (Wösten *et al.*, 2006).

Less than 4 percent of tropical peat swamp forest in Southeast Asia remains in a near-intact condition, with 37 percent classified as degraded forest (logged-over), 24 percent as deforested, burnt or both, and 32 percent as under agriculture (Miettinen and Liew, 2010). Degradation of peat swamp forest leads to the loss of most, if not all, ecosystem services, including carbon storage and hydrological regulation (Page *et al.*, 2009). Total annual CO_2 emissions from Indonesian peatlands (from peat oxidation, fire and loss of biomass) over the period 2000 to 2006 have been estimated at 640 Mt of CO_2, equivalent to 2.1 percent of current annual global fossil fuel emissions (Bappenas, 2010).

Given the extent of degraded peat swamp forest in Southeast Asia and the implications for increased carbon emissions to the atmosphere, there is an international effort to promote ecosystem rehabilitation. The Indonesian Government is collaborating internationally to initiate large-scale restoration programmes on degraded peatlands.

Overcoming the barriers to the restoration of tropical peat swamp forest

When intact tropical peat swamp forest is disturbed, environmental effects include changes in microclimate (higher temperatures, reduced humidity, altered light levels); lowering of the water table, which in undisturbed peat swamp forest is close to or above the peat surface; reduction or loss of the hummock–hollow peat surface to-

Figure 14.2.
The distribution of the main peat deposits in Southeast Asia (shaded areas). Most peatlands occur on the islands of Sumatra and Borneo (Kalimantan, Sarawak and Brunei) and in peninsular Malaysia.

Source: derived from Miettinen *et al.* (2012).

pography, and hence loss of hummock surfaces on which tree seedlings establish; and increased peat oxidation and fire occurrence, both resulting in peat surface subsidence and increased risk of flooding (Page *et al.*, 2008).

Disturbance history will be unique to each location and will influence the rate and type of forest regeneration. Altered hydrological conditions and fire are the main barriers to forest regeneration (Page *et al.*, 2009); therefore, a preliminary assessment should be undertaken of land cover, drainage history (time of installation, location, size of drainage features) and fire history (fire location, frequency and severity). If anthropogenic impacts are minimal, forest regeneration capacity is likely to be relatively high. For example, in selectively logged forest or sites subject to a single, low-intensity fire, secondary peat swamp forest can establish over approximately ten years (Hoscilo *et al.*, 2011). With increasing intensity of degradation (e.g. following intensive drainage and frequent and/or severe fires), regeneration of woody species is limited or entirely suppressed and species diversity greatly reduced (Wösten *et al.*, 2006). Thus, active restoration measures will be required to facilitate reforestation. In addition, information on annual hydrological variation (duration and extent of low and high water levels, which will influence survival of transplants, especially at the critical seedling stage), natural seed dispersal by mammals and birds, levels of competition from invasive, non-woody species (ferns, sedges) and availability of nutrients and mycorrhizal fungal symbionts in the surface peat, all need to be assessed as potential barriers to tree regeneration. Active barriers identified in recent studies include wet-season flooding, high light intensity, low levels of mycorrhizal fungi in the surface peat and low levels of seed dispersal (Page *et al.*, 2008; Graham and Page, 2012).

Species selection

In Southeast Asia, peat swamp forest supports at least 1500 plant species (Posa, Wijedasa and Corlett, 2011). Of these, approximately only 11 percent are endemic to peat swamp forest, although all are adapted to the wet peatland environment. In stark contrast, only two or three pioneer woody species occur on heavily degraded sites (Page *et al.*, 2009). These typically include wind-dispersed species (e.g. *Combretocarpus rotundatus* (Miq.) Danser), species dispersed by small animals, (e.g. *Cratoxylon glaucum* Korth., *Syzygium* spp.) or fire-resistant species (e.g. *Melaleuca* spp.).

There is little formally published literature on the selection of appropriate species for restoration of tropical peat swamp forest, but a reasonable wealth of grey literature, including conference proceedings and internal project reports. The Kalimantan Forest Climate Partnership, a tropical peatland restoration demonstration project supported by the United Nations Collaborative Programme on Reducing Emissions from Deforestation and Forest Degradation in Developing Countries (REDD) and developed under an Australia-Indonesia Government partnership, has pooled available literature into a silvicultural review of peat swamp forest tree species occurring in Central Kalimantan. This is an important stage in identifying potential transplant species for this region while also highlighting gaps in silvicultural knowledge that need to be addressed in order to develop restoration best practice (Graham, 2009). This is the first step in identifying species with appropriate ecological traits for use across the range of environmental conditions characteristic of degraded tropical peatlands.

To date, a small proportion (< 5 percent) of the peat swamp forest tree species have been used in restoration trials; frequently used species are *Shorea balangeran* Burck, *Alstonia spathulata* Blume and *Dyera polyphylla* (Miq.) Steenis. The last of these is popular because the bark can be tapped for latex, which provides a source of income for local communities. Ideally, species to be planted should be selected based on their ecological tolerance of site-specific conditions, but since there is limited autecological knowledge of the vast majority of tropical peat swamp forest tree species, restoration efforts have proceeded largely by trial and error. Peat swamp forest that has been logged and drained is exposed to high light levels and drought during the dry season, but does not suffer from flooding. Other sites, such as those closer to waterways or where the peat surface has subsided as a result of peat oxidation and fire, may flood regularly (Wösten *et al.*, 2006). Some peat swamp forest tree species have a broad tolerance of this range of degraded conditions; *Shorea balangeran* and *Syzygium* spp., for example, are species of riverine forest at the edge of the peat dome and tolerant of both high light and high water levels, making them suitable species for planting on open areas subject to flooding. In contrast, *Tetramerista glabra* Miq., a tree of mixed swamp forest on deeper peat, will grow in degraded areas but only if there is adequate shade, and thus is best suited to enhancement planting or planting at the forest edge. Other species showing high survival rates in restoration trials include *Combretocarpus rotundatus, Koompassia malaccensis* Benth., *Melaleuca cajuputi* Powell, *Syzygium oblatum* (Roxb.) Wall. ex A.M.Cowan & Cowan and *Tetramerista glabra.* Nursery seedlings propagated from seed or wildlings are commonly used as propagation material. Seedlings are normally cultivated in nurseries for at least six months before planting out on the degraded areas.

In practice, the greatest risks to the long-term success of restoration trials on tropical peat swamp forest to date have proved to be prolonged deep flooding and fire rather than poor choice of species for planting. Efforts to overcome seedling mortality during periods of high water level have included constructing artificial planting mounds, while reducing fire damage requires active fire prevention during the annual dry season when drained peatlands are at greatest risk of ignition (Giesen, 2009; Page *et al.*, 2009).

Community knowledge and participation

Local communities can provide knowledge of site history and insights into the social impediments to forest restoration. A study in Central Kalimantan that employed community participation, focus groups and interviews investigated the understanding and attitudes of local people relating to forest degradation, their current dependency on and attitudes towards the forest and their hopes for its future (Graham, unpublished data). The study highlighted important issues that would need to be addressed in a local forest-restoration action plan, and demonstrated the wealth of forestry and ecological principles understood by the community, their desire to be involved with restoration activity and their insights on how transplanted tree species should be selected and used.

Concluding remarks

Tropical peatland degradation is now widely accepted as a matter of international concern, and restoration is seen as essential. Key stages necessary to achieving restoration in topical peatlands include improved knowledge of how to tackle fire management, hydrological rehabilitation and species selection and better assessment of the diverse array of secondary regeneration barriers. Knowledge gained from long-term monitoring of restoration sites will also be critical to development of pathways to more efficient landscape-scale restoration, as will be effective local community engagement.

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14.3. Support to food security, poverty alleviation and soil-degradation control in the Sahelian countries through land restoration and agroforestry

David Odee and Meshack Muga

Kenya Forestry Research Institute

Sahelian countries are severely affected by drought and desertification, with a significant southward expansion of the desert into gum- and resin-producing zones. Since the 1970s countries within the sub-Saharan Sahelian region have experienced droughts that adversely affect livestock production, agriculture and woodlands. A regional project implemented in 2003–2010 applied a coordinated strategy for restoration of degraded lands to support agrosilvipastoral activities. The development objective of the project was to contribute to sustainable development, food security and the fight against desertification through the promotion and integration of gum and resin production into rural economic activities in Africa. The immediate objective was to strengthen the analytical and operational capacity of six pilot countries (Burkina Faso, Chad, Kenya, Niger, Senegal and the Sudan) to address food security and desertification problems through the improvement of agrosilvipastoral systems and the sustainable development of the gum and resin sectors.

Site information

The project sites were located mainly in degraded silvipastoral sites in arid and semi-arid ecological zones of sub-Saharan Africa. Restoration plots ranged from two to 850 hectares, with a total area of approximately 13 000 ha. Location of the sites and the landscape context in each country were as follows:

- Burkina Faso: 27 sites (no site information).
- Chad: degraded land with low shrub and tree cover with natural indigenous species.
- Kenya: low isolated shrubs of natural *Acacia* and *Commiphora* species in Marsabit District (Merille, Logologo and Laisamis sites), and in the southern rangelands in Makueni District (Kibwezi and Kiboko sites).
- Niger: natural woodlands occurring in the Departments of Mirriah, Aguié, Madaoua, Say, Kollo and Plateau de Kouré.
- Senegal: isolated acacia woodlands in the main administrative regions in Louga, St Louis, Matam, Tambacounda and Diourbel.
- The Sudan: in the *Acacia senegal* belt and sites adjacent to forest reserves in North Kordofan, Sennar and Blue Nile states.

Site conditions prior to restoration activities were dry, open and highly degraded, with isolated shrubs or woody species; some sites were devoid of vegetation. Key sources of livelihoods for the local communities were landscape restoration for livestock grazing, agroforestry and provisioning of other ecosystem services such as fuelwood and soil fertility improvement. Gum production was the strategic income-generating activity and driver for restoration activities. There was no involvement in carbon offset schemes or activities targeted at reducing emissions from deforestation and forest degradation (REDD+), despite their strong relevance.

Restoration activities

A participatory approach was adopted by the project, with high interest and ownership shown by local communities. Sites for restoration were selected through participatory approaches with local communities. Preliminary surveys and collection of baseline information helped to identify target groups. Main interest groups for the project were local communities, local government departments for environment and agriculture, FAO, the Network of Natural Gums and Resins in Africa (NGARA) and the Inter-Departmental Working Group on the Convention to Combat Desertification.

Restoration activities were initiated in 2004. The project employed a mechanized water-harvesting technology (the Vallerani system) that digs microbasins while ploughing degraded soils, and used this to develop acacia-based agrosilvipastoral systems and reverse land degradation. The technology has good potential to harvest and store water for associated vegetation (trees and crops) long after rainfall. The mechanized system is efficient and can be used to prepare swathes of degraded lands within a shorter time than can be achieved with local traditional systems, which employ hand hoes or oxen-powered ploughs. This system has been used successfully in rehabilitation of degraded lands in North Africa.

Native tree species were preferred because of their adaptation to the local environment, availability of planting material and awareness and use of the species by local communities. The key native species used across the sites was the gum arabic tree, *Acacia senegal* (L.) Willd., because of the commercially valuable gum it produces and thus the potential for income generation. In addition to producing gum, it is a nitrogen-fixing tree, enriching the soil through the nitrogen-rich litter it produces, producing fodder during the dry season and stabilizing the soil. Other species used were:

- the drought-tolerant timber tree species, *Melia volkensii* Gürke (used at Kibwezi and Kiboko sites, Kenya)
- *Acacia seyal* Delile, *Acacia nilotica* (L.) Delile, *Bauhinia rufescens* Lam. and *Ziziphus mauritiana* Lam. in Niger
- *Acacia mellifera* (M.Vahl) Benth. in Senegal
- *Acacia seyal, Acacia mellifera, Acacia nilotica* and *Albizia* spp. in the Sudan.

Some exotic or non-autochthonous species were also used in agroforestry systems, namely *Jatropha curcas* L. (biofuel) and *Mangifera indica* L. (mango) for income generation in Kenya and *Azadirachta indica* Adr. Juss. for medicinal purposes and fodder in the Sudan.

Seeds of *Acacia senegal* were collected from local populations or the nearest possible sources using standard procedures to ensure genetic diversity. Seedlings were raised in local or community-based nurseries. In Niger, *Acacia senegal* seeds from Kordofan provenance were also used but it is not clear in which site they were planted.

Acacia senegal trees were planted in rows along the microbasins with an interrow spacing of 4–6 m. Annual crops such as millet and maize were planted between the tree rows to create agroforestry systems. The plots were located near villages and ranged in size from five to 100 ha.

Monitoring and experiences gained

Management and monitoring of the trees were carried out by the local communities. Trees were maintained and intercrops planted annually during rainy seasons. Performance assessment was based on survival of seedlings and growth and yields of annual intercrops. Seven years after planting, *Acacia senegal* and other species were reported to have survived and established successfully, but no statistics are available.

The overall conditions of the sites are also reported to have been restored, including improved soil health and fertility and vegetation cover. The local communities are using the restored land for agroforestry activities. Agricultural intercrops include sorghum, beans, pigeon pea, cowpea, green gram, pearl millet, watermelon and maize. The local communities are in accord with the need to adopt an integrated approach to addressing the problems of land degradation and poverty. Technological aspects of production and processing should be linked to marketing, while livelihood diversification should be introduced to ensure sustainability. The participating countries were highly receptive to the experiences of rehabilitating degraded lands and rationalizing the production of gums and resins as the key driver to restoration in the drylands.

Overall, the restoration activity was deemed a success, considering the good establishment and survival of the trees planted, as well as integration and adoption of agroforestry activities within the restored areas by the normally pastoral communities. Problems encountered included prolonged droughts, especially during planting and establishment, damage to planted trees by livestock and wild animals, and the communal land-tenure system, which affects ownership and responsibility over restored areas. Income-generating activities should be identified, promoted and supported to offer incentives to the local communities to backstop the sustainability of the restoration activities.

There are plans to upscale the restoration programme, and to link to new initiatives such as the Great Green Wall for the Sahara and Sahel initiative supported by the African Union and the European Union.

14.4. The use of native species in restoring arid land and biodiversity in China

Lu Qi and Wang Huoran

Chinese Academy of Forestry, Beijing, China

Generally, desertification is mainly caused by climate change and human activities. Overgrazing, inappropriate agricultural use and excessive collection of fuelwood and medicinal herbs reduce vegetative cover, rangeland capacity and biological diversity, and expose the soil to erosion (CCICCD, 1997). China is one of the countries worst affected by desertification. The areas most seriously affected are found in the northern, northeastern and northwestern parts of the country. The Chinese Government, scientists and local communities have made great efforts to restore land and biodiversity affected by desertification.

The climate and soils of northwestern China pose extremely severe challenges to the development of vegetation. The climate is characterized by frequent, strong winds, very low rainfall and high evapotranspiration. Sandstorms are destructive and hamper agriculture, forestry and animal husbandry. Most of the arid areas experience a typically continental climate, with annual precipitation less than 250 mm. Temperature changes dramatically, daily, seasonally and spatially; for example, the maximum temperature recorded in the Xinjiang region is 47.5 °C in Turpan and minimum temperature is –43 °C in the Zunger Basin. Desert soils are relatively undeveloped. Soil texture is usually coarse sandy loam, sand and gravel. Organic matter and nitrogen contents are low, but abundant soluble mineral salts are accumulated at the soil surface, resulting in high pH and, in many cases, large areas covered by saline crusts.

Plant genetic resources in the arid areas of China

It is estimated that there are over 1000 species of trees and shrubs in the arid and semi-arid areas in northern China (Li, 1992). Many of these, especially the shrubs, have been used for environmental improvement, dune stabilization, afforestation, and production of food and other human requirements.

A number of tree species have been used in dune plantings in the arid and semi-arid regions of China. *Pinus tabuliformis* Carrière, *P. sylvestris* var. *mongolica* Litv., *Platycladus orientalis* (L.) Franco, *Sabina chinensis* (L.) Antoine, *Juniperus rigida* Siebold & Zucc. and *Picea crassifolia* Kom. are native coniferous species adapted to a cold and dry environment with annual rainfall of less than 400 mm (Zhao, 2005). *Picea crassifolia*, for example, is naturally distributed up to the treeline and is critical for watershed management in the glacier-covered Qilianshan Mountains in Gansu Province. *Sabina chinensis* and *S. vulgaris* Antoine are useful for ground cover to protect soil from erosion on harsh sites.

Only a few indigenous hardwoods can survive in the harsh environment of northwestern China. *Populus simonii* Carr. and *P. alba pyramidalis* (Bunge) W.Wettst. are mostly used for establishing windbreaks or shelterbelts on agricultural land or around oases (Zhao, 2005). Of the species used in shelterbelts established in the 1970s in northwestern China, only *Ulmus pumila* L. survived drought and insect pests. Other hardwoods, such as *Sophora japonica* L., *Salix matsudana* Koidzumi and *Ailanthus altissima* (P. Mill.) Swingle are also often planted as shade trees around villages in northwestern China.

The diversity of shrubs is, however, relatively richer than that of trees (SSG, 2003). Efforts to maintain biological diversity and reclaim land have in recent years increasingly used shrub species as well as trees. Shrub communities have important ecological functions in conservation of soil, water and biodiversity. Most indigenous shrubs are better adapted to dry soil, poor nutrient availability and temperature extremes than are trees. Moreover, several shrubs have high economic value as medical plants (e.g. *Sophora alopecuroides* L. and *Lycium chinense* P. Mill). There is great potential in the utilization of the genetic resources of shrub species evolved in the arid and cold environment in northern China.

A number of shrub species have been used to stabilize sand dunes and for other ecological and economic uses. *Caragana korshinskii* Kom., a leguminous shrub, is used extensively for dune stabilization, bioenergy, fodder, and soil and water conservation in arid and semi-arid areas in the middle reaches of the Yellow River. It is xerophytic, capable of prolific coppicing and tolerant of animal browsing. It is native to northwestern China, where it grows in areas with annual rainfall of approximately 200 mm and air temperatures ranging from 70 °C above the sandy ground in summer to –30 °C in winter. Other species in the genus *Caragana* with similar biological traits, such as *C. intermedia* Kuang & H.C. Fu and *C. microphylla* Lam., are also commonly used for land reclamation and dune stabilization (Li and Bao, 2000). *Hedysarum scoparium* Fisch. & C.A. Mey. and *H. mongolicum* Turcz., also leguminous shrubs, are more adapted to moving dunes, semi-fixed land or fixed sandy land in areas with annual rainfall between 150 and 300 mm. They are tolerant of wind erosion, with deep root systems and strong coppicing ability. *Hedysarum mongolicum* colonizes sites by producing thick suckers around main stems. *Glycyrrhiza uralensis* Fisch. (Chinese liquorice) is a perennial herb or semi-woody shrub of the Paplionaceae family that has been successfully grown in the desert, providing a good income to farmers as a traditional Chinese medicinal plant.

Ammopiptanthus mongolicus (Kom.) S.H. Cheng is unusual in that it is an evergreen species in the cold desert area. It forms symbioses both with nodule-forming bacteria and mycorrhizal fungi, which together help it fix nitrogen and take up other nutrient elements in the soil. This species grows well in soil with pH values of 8 to 9 (SSG, 2003). *Tamarix chinensis* Lour., *Hippophae rhamnoides* L., *Elaeagnus angustifolia* L., *Salix psammophila* Z.Wang & Chang Y.Yang, *Haloxylon ammodendron* (C.A. Mey) Bunge ex Fenzl and many other species are used to stabilize moving sands and establish shelterbelts as understorey or living hedges. Research in genetic improvement with *H. rhamnoides* has been carried out over the last 30 years and good seed resources have been established and utilized (Lian and Chen, 1992; SSG, 2003).

Methodology of tree planting in arid areas

Generally, three approaches are used to establish plantations of woody plants in China (Zhao, 2005). The first approach is to plant trees by hand or machine. This is commonly used to establish plantations, especially for shelterbelts. Seedlings or cuttings used are normally of a large size; for instance, it is quite common to use cuttings of poplars over 2 metres in length. The seedlings or cuttings may be watered and fertilized at planting time to assist survival. The second approach is to sow seed by airplane. This is the usual method when dealing with large areas of sand dunes or wild lands, usually during the rainy season. Timing is critical for aerial sowing. In the west of northeastern China and the Loess Plateau, plantations of pines, larches or shrubs are sometimes established by this method. Approximately 30–50 percent of the seeds sown germinate and establish (Qi, 1999). The third method is to close a mountain area or sandy land to human use or to mitigate animal pressure. This allows natural vegetation to re-establish and the land to recover. It takes many years to restore vegetation this way in the arid and semi-arid areas, but the approach requires little in terms of direct inputs. This concept of land management has been used over a long period and has been shown to be efficient and economical (Zhao, 2005).

Prior to planting, sites are usually prepared to reduce runoff of water and increase infiltration.

Many different methods have been developed and used depending on the situation. However, two methods of site preparation are widely used. On shallow slopes or relatively flat sites a 1 × 1 m square pit is dug to a depth of 20–30 cm and a seedling is planted in the centre. On steeper slopes or stony sites a fish-scale-shaped pit, or level terrace, is made and a seedling is planted at the lowest point of the pit. On sandy land or mobile dunes seedlings or cuttings are planted on unprepared land.

Straw checkerboards and barriers have been shown to improve establishment of trees or shrubs planted on dunes (Fan, Xiao and Liu, 1999). Newly planted seedlings or cuttings are easily damaged by wind or buried by sand. To prevent this, artificial barriers and straw checkerboards are established prior to planting trees in order to reduce wind velocity near the ground and slow down the movement of sand. Sand barriers can also prevent sand from drifting inside fixed-sand areas. Wheat or rice straw for making checkerboards is readily available in rural areas. Artificial sand barriers are usually made with shrub stems or other similar materials. Trees are planted in each grid of the checkerboards. Plantations usually take two or three years to establish after planting; the straw checkerboards and artificial barriers help stabilize dunes while the trees are establishing. This method has been widely used for dune plantings.

Recently, drip irrigation systems have been more widely used to maintain plantations in extremely dry areas, in some cases where annual precipitation is not more than 400 mm (Xu *et al.*, 1999). Such systems are very expensive to set up, but are highly efficient and save water compared with traditional irrigation systems. Drip systems have, for example, been used in the roadside planting project along the desert highway in Takelamagan, Xinjiang Uyghur Autonomous Region, and a plantation project in the vicinity of Yinchuan City, Ningxia Hui Autonomous Region. Research is also being carried out to identify plant species and accessions with lower water requirements. Competition for water between trees and agricultural crops must be minimized or avoided if possible.

Figure 14.3.
Planting methods (left) and species (right) used in arid land restoration in China

Lessons learned from the past

Species selection is a key to the success of biodiversity recovery or land reclamation. In the past, too much emphasis was given to planting tree species in monocultures, which led to poor adaptation and problems with pests and diseases. One or very few clones of poplars have commonly been planted across landscapes without having tested the clone prior to mass planting. Some poplars, such as *Populus simonii* and *P. alba*, even though native to northwestern China, are not suitable for areas receiving less than 400 mm annual rainfall, yet shrub communities develop well under such conditions. More shrub species are now being used to stabilize dunes, for bioenergy production and for conserving soil and watersheds in the dry areas in northwestern China. Another change in approach is to use more fruit and nut trees to support the market economy. Chinese chestnut (*Castanea mollissima* Blume), Siberian apricot (*Prunus armeniaca* L.) and pear (*Pyrus betulaefolia* Bunge), for example, are tolerant of harsh environments and are able to produce nuts and fruits under such conditions (SSG, 2003; Zhao, 2005).

Insect damage has in recent years drawn more attention to the management of dune plantations, since it has become more serious in the dry environment in northern China. Species diversification is effective in controlling insect damage: a single species or clone must not be planted over large areas. Poplars are easily damaged by a number of insects and diseases, such as the Asian long-horned beetle (*Anoplophora glabripennis*), poplar grey spot (*Coryneum populineum*) and poplar leaf rust (*Melampsora laricis-populina*) (FAO, 2007). Compared with poplars, *Ulmus pumila* (Manchurian elm) and *Salix matsudana* are attacked much less by insects. In some counties of the Ningxia Hui Autonomous Region, among shelterbelts that were established in 1970s, only those established with elms remain. From an ecological point of view it is better not to establish windbreaks using tall tree species. Shrubs such as *Caragana* spp. and *Hippophae rhamnoides* would be a good alternative.

Grazing animals and fuelwood collection pose problems in the management of dune plantations, but this can be addressed by increasing the awareness of local residents of the environmental importance of shelterbelts and other plantings on dunes.

Changes in land tenure in recent years have implications for land restoration work in northern China. In the past, windbreak systems were established in very regular networks extending over long distances. However, this is no longer possible because agricultural land has been divided into small lots, each owned by an individual. Under the new system farmers are entitled to grow any tree species they wish. Choice of species is mostly driven by market demand. The government can lead planting trends only by adopting appropriate policies and providing technical and financial help.

Greenness, a new concept to express coverage of vegetation in a given geographic area, has been proposed as a replacement for the concept of forest coverage. Greenness can be calculated by remote sensing from satellite images. It is argued that greenness is a better measure of vegetation status than forest coverage since forests cannot establish or develop well in the arid and semi-arid areas, and shrub or grass communities also have environmental significance. This conceptual shift will lead to change in the strategy of land reclamation, especially in terms of species composition and shelterbelt or stand structure. Disturbed shrub communities and grassland will receive more attention for conservation and management in arid areas (Shan, 2007).

Research gaps identified

It is foreseen that environmental issues, especially those related to climate change, will become more serious in the near future, affecting human living conditions and social and economic development. In particular, sustainable social development can be expected to be severely constrained by the decline in availability of biological resources. Obviously, arid and semi-arid areas do not hold potential for the development of commercial forestry, and therefore the focus should be on rehabilitation and sustainable management of arid land resources through biological approaches. Research and development in the conservation of biodiversity and land reclamation should emphasize the exploitation of genetic resources of native species, especially shrubs and perennial herbs with economic values. Water availability is a critical factor for species survival, and plant species' water-saving strategies should receive more attention in research programmes.

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14.5. Using native shrubs to convert desert to grassland in the northeast of the Tibetan Plateau

Yang Hongxiao[1] and Lu Qi[2]

[1] College of Resources and Environment, Qingdao Agricultural University, China
[2] Institute of Desertification Studies, Chinese Academy of Forestry, China

The Tibetan Plateau lies at an average of more than 4500 m above sea level. The climate is much colder and drier on the northern slope than on the southern one. Mean annual temperature varies from –5.8 °C to 3.7 °C and annual precipitation is usually less than 400 mm. Such conditions adversely affect plant growth and maintenance of vegetation. Adding to this, overgrazing and poor farming practices have destroyed the vulnerable vegetation. Desertification of former grassland intensified in recent decades and secondary deserts increased. Secondary deserts look like pale scars on the grassland, where the sand is bare and exposed to the wind. When strong winds blow, such sand moves easily to form sand drifts and create sand storms. The government and local people are anxious to revegetate these degraded areas to improve environmental health and support sustainable development. However, this is not easy; drifting sand restricts plant colonization. Fortunately, researchers have found that native shrubs can help solve this problem (Yang *et al.*, 2006).

Research results

Shazhuyu is a village in the northeast Tibetan Plateau (100°13′53.97″E, 36°14′08.41″N), with annual precipitation of about 280 mm and mean annual temperature of about 5 °C. There are many patches of secondary desert around the village. Experiments were started in 1958 to investigate ecosystem restoration. These demonstrated the following:

- The secondary deserts are restorable because the climate is not as dry as in primary deserts, such as Taklimakan in Xinjiang, China.
- Frequent sand drift driven by strong wind constrains the process of grass colonization and ecosystem restoration.
- Transplanting native shrubs into the secondary desert patches reduces sand drift, allowing grass to quickly colonize the area.
- With the help of the shrubs, grass communities develop smoothly over a period of several years.
- Once the grass communities are strong enough to persist by themselves, the shrubs begin to disappear from well-restored ecosystems that have become grassland with high vegetation cover.
- The grassland can persist once the shrubs have gone because most of the grasses are perennial, stabilizing the sand with their roots, stems and leaves even in winter.
- Upon reaching this stage, the ecosystem can tolerate some grazing.
- If grazing is well managed, desertification need not occur.

Thus, native shrubs can play an important role in converting secondary desert to grassland (Yang *et al.*, 2006; Lu *et al.*, 2009).

Practical application

Artemisia ordosica Kraschen. and *Caragana korshinskii* Kom. are two common species of native shrubs in northwest China, mainly appearing in arid deserts or on sandy land in the northeast of the Tibetan Plateau and the west of the Mongolian Plateau (Yang *et al.*, 2006; Lu *et al.*, 2009). Shazhuyu falls within their natural distribution, and they have been found to grow well in secondary deserts in the area. Researchers in charge of ecosystem restoration selected them as the main species to stabilize sand and facilitate ecosystem restoration. Biologically, they are well adapted to desert conditions. Nevertheless, they spread through the deserts very slowly. Harmful sand drift greatly slows the spread of *A. ordosica* populations, although their tiny seeds are readily dispersed by wind. Spread of *C. korshinskii* is limited by the plant's heavy seeds, which are not dispersed by wind (Yang *et al.,* 2009). The slow natural spread of the two species limits their contribution to ecosystem restoration. To overcome this, researchers developed a method to transplant saplings or mature individuals of *A. ordosica* and *C. korshinskii* or to sow their seeds into bare deserts. Researchers also adopted two other methods using enclosures and mechanical barriers (Lu *et al.,* 2009). Enclosures protect target deserts from grazing, while mechanical barriers made of clay, dry straw, dry twigs or rock fragments prevent sand from drifting. The two methods significantly increase the survival rate of *A. ordosica* and *C. korshinskii*. To some extent, they can also facilitate grass colonization on the deserts, but this effect is not as prominent as that of mature shrubs.

Application to arid-ecosystem restoration

Using these methods, researchers have succeeded in converting some sand deserts back to grassland dominated by the perennial grass *Leymus secalinus* (Georgi) Tzvel., which is palatable to yaks, sheep, cattle and horses. This conversion takes about 50 years. In contrast, untreated areas are still bare and consist of mobile sand, showing no signs of restoration.

Experiences from Shazhuyu set a good example for other arid regions in the Tibetan Plateau and elsewhere, where native shrubs play an important role in restoring secondary deserts. Throughout the Tibetan Plateau, native shrubs that adapt well to a sandy environment include *A. ordosica*, *C. korshinskii*, *Artemisia xigazeensis* Ling et Y.R. Ling, *Sophora moorcroftiana* (Benth.) Baker, *Tamarix ramosissima* Ledeb., *T. hohenackeri* Bunge, *T. laxa* Willd., *T. elongate* Ledeb., *Hippophae rhamnoides* Linn., *Nitraria tangutorum* Bobr., *N. sibirica* Pall., *Lycium ruthenicum* Murray, *L. dasystemum* Pojarkova, *L. cylindricum* Kuang, *Haloxylon ammodendron* Bunge and *Ephedra przewalskii* Stapf (Chen and Liu, 1997; Liu, Gao and Jiang, 2003). Different regions will have different native shrubs, and local people should be asked to recommend the most promising for ecosystem restoration in their locale.

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14.6. Reforestation of highly degraded sites in Colombia

Luis Gonzalo Moscoso Higuita

FORESTPA S.A.S., Colombia

The method presented here was developed by the author based on observations of natural succession taking place on land that was abandoned by rural communities as a consequence of armed conflicts. Study sites included land that had been highly degraded by agricultural practices, cattle breeding and mineral extraction. Natural succession was thoroughly studied on the sites, particularly with respect to the dominance of certain species, the abundance and diversity of weeds in the understorey, the formation of soil organic matter, the appearance of micro-organisms and recolonization by natural fauna.[25]

Based on these observations, the author developed strategies to stimulate natural processes relating to wind-, water- and animal dispersal of seeds. This involved selection of the site, establishment and maintenance of fences, selection of elite mother trees as candidates for future seed collection, propagation in transitory nurseries, and establishment and maintenance of plantations. Full details of the approach are presented in Moscoso Higuita (2005). So far, this method has been applied on 1292 ha of land, of which 751.79 ha are degraded by gold mining, intensive cattle-raising and unsustainable cultivation of agricultural crops in the Colombian municipality of Cáceres, Antioquia.[3]

Methodology

Various forest restoration strategies were developed in accordance with the type and degree of soil degradation, including intensive and extensive cattle-raising, unsustainable agricultural practices and opencast gold mining. For this reason, the methodology is initially described through a two-track approach, whereby some common themes such as social and administrative aspects are described jointly.

The starting point for the development of this methodology involved the traditional knowledge of the ethnic groups living in the project area. Colombia is a highly diverse country, both in terms of biota and societies of different origins, including Afro-Colombians, indigenous people, mestizos and mulattos. Valuable traditional knowledge and practices were combined with external technical and administrative knowledge, always in a context of respect and equity in terms of goods and services. Second, it was necessary to know what environmental resources were present in the buffer zone surrounding the degraded area. This involved selection of target species, understanding existing opportunities to connect vegetation fragments, identification of superior or elite mother trees for collection of germplasm, and assimilation of basic information on soils, vegetation, hydrology, fauna and micro-organisms, among other aspects. Once sufficient knowledge had been collected to allow characterization of both the degradation state and the available natural resources, the forest restoration strategy was put into practice.

The project area was divided into zones according to prevailing conditions of soil degradation using tools such as aerial photography and maps covering several decades. This was done to get a better understanding of the sites' history.[26] In

[25] The Cáceres I and II projects, quoted as a case study, account for 751.79 hectares, located at 7°34'38.6"N, 75°19'47.4"W, and are characterized as tropical rainforest according to the Holdridge classification.

[26] Maps and aerial photographs were obtained from Instituto Geográfico Agustín Codazzi (IGAC).

addition, a topographic survey was undertaken with the help of a GPS to determine the location of all landforms, including paths, roads, forest relicts, crop residues, constructions and others. Based on this information, the landscape was divided into plantation areas that were isolated from one another by means of living hedges consisting of fast-growing and vigorously rooting plant species (planted by means of vegetative reproduction, and as grass seedlings) and barbed-wire fences.

Soil samples were collected and characterized chemically and physically with the help of specialized laboratories. Based on these analyses, companies producing organic fertilizers prepared a special fertilizer mix based on compost to restore nutrient deficiencies (Table 14.1). Fungi and bacteria were added to the soil to promote biodegradation of organic materials and increase availability of nutrients. Ultimately, this is expected to lead to an improved nutritional balance and improved soil-water retention.

TABLE 14.1.
Composition of fertilizer mix applied

Component	Percentage of content
Chicken manure	40.0
Mushroom compost	20.5
Mycorrhizae	18.0
Phosphoric rock	10.0
Gypsum	7.0
Magnesium sulphate	3.0
Agrimins	1.0
Humiplex 50G	0.5
Total	**100**

Standard protocols were followed to identify good seed sources and planting material for production of seedlings in transitory nurseries. Selection was based on:

- research conducted at other locations with similar edaphic and climatic conditions;
- volumetric efficiency of the species;
- nutritional requirements of the species;
- geographic distribution of the species;
- adaptive capacity of foreign or exotic species;
- occurrence of the species in the region;
- economic, cultural and ecological value;
- ease of accessibility of seeds, high germination percentage and broad genetic base; and
- resistance to pests and diseases.

In the Cáceres II project, plantations were established with seeds from seed orchards and from elite trees selected from areas near the restoration site and from other locations that were part of the species' natural distribution. A register was assembled of every elite tree, containing details of its morphology, phytosociology and health, and the location and topography where it was found. For each of the selected species, protocols were followed for germination and growth in transitory nurseries. Planting bags were filled with a mix of mineral soil, fine sand and the fertilizer mix detailed in Table 14.1, in a ratio of 3:1:1. The substrate was further enriched with sprays composed of fungi and bacteria that were produced and supplied by a laboratory specialized in biotechnology. The bacteria included in these sprays belonged to the genera *Bacillus, Pseudomonas, Azotobacter, Azospirillum, Beijerinckia, Nitrosomonas, Nitrobacter, Clostridium, Thiobacillus, Lactobacillus* and *Rhizobium.* These are among the most important bacterial genera involved in the degradation of organic and inorganic compounds, and hence promote availability of nutrients to plants. The most important genera of soil actinomycetes used for enhancing plant nutrient uptake were *Streptomyces, Nocardia, Micromonospora, Thermoactinomyces, Frankia* and *Actinomyces.* Finally, mycorrhizal fungi from the genera *Rhizophagus* (ex *Glomus*), *Acaulospora, Entrophospora* and *Gigaspora* (which enter in symbiosis with the roots; Delgado Higuera [1999]) were added to the substrate. The same substrate was used in the planting bags, while transplanting the germinated plants, and as fertilizer in the established plantation.

Light vehicles such as pickup trucks were used to transport plant material to the planting sites. Within sites, seedlings were transported on the backs of horses and mules to reduce soil compaction. Sites were weeded manually or, in some cases, using mowers, especially on sites with a grass cover. During weeding some isolated trees were left standing and were pruned; these act as natural seed sources, attract dispersal agents and serve as biological corridors. Bordering groves, marshes, streams, creeks and rivers were left untouched. This approach is categorized as enrichment; a combination of sparing interesting species already available on site during initial weeding activities and planting new trees in clearings.

The planting design varied according to the number and distribution of trees already present in the sites. On land without any remaining trees, seedlings were planted at a spacing of 3 × 3 m. At sites that were previously used for livestock ranching and agriculture, holes of 0.4 × 0.4 × 0.4 m were dug and filled with crumbled soil to counter soil compaction. In soils degraded by alluvial gold mining, holes were dug up to 70 cm deep to reach deeper, less or non-polluted soil layers. On sites with a grass cover, areas of 1-m diameter around each plantlet were weeded. In soils that were polluted from mining this was not necessary as there was generally no vegetation left. Soil from each planting hole was mixed with 120 g of fertilizer mix (Table 14.1), 5 g of Terracotem (soil conditioner with hydro-retainer) and organic–mineral fertilizer. Seedlings that failed to establish were replanted 45 days later, but mortality did not exceed 10 percent.

Maintenance depended on the nature of the weeds emerging. Naturally regenerating tree, shrub or herbaceous plants of interest were identified and left standing. Degree of interest was based on wood quality, proximity to the plantlets, ecological value, growth, abundance, incidence and presence in the plantation. On sites with lower vegetation cover, regeneration was assisted or promoted by dispersing seeds of pioneer species (seed collected from marked trees, seed stands and clonal seed orchards). Assisted regeneration refers to methods that enhance the *in situ* regeneration of plant species, either by promoting the health and growth of plants already available on site or by distributing seeds of desired species. The species that were initially established in the Cáceres II project were acacia and teak (*Acacia mangium* Willd. and *Tectona grandis* L. f., both exotic species), Colombian mahogany (*Cariniana pyriformis* Miers), red cedar (*Cedrela odorata* L.), roble (*Tabebuia rosae* DC), Brazilian firetree (*Schizolobium parahyba* (Vell.) S.F. Blake), amami gum (*Hymenaea courbaril* L.), choiba or almendro (*Dipteryx oleifera* Benth.), melina (*Gmelina arborea* Roxb., another exotic), downtree (*Ochroma pyramidale* (Cav. ex Lam.) Urb.), Spanish elm (*Cordia gerascanthus* L.), Devil's ear (*Enterolobium cyclocarpum* (Jacq.) Griseb.) and ceiba tolúa (*Bombacopsis quinata* (Jacq.) Dugand), accounting for a total of 359 830 individuals.

In the 190 monitoring sites that have been installed so far using this technique, 122 tree species in various families and genera have been shown to have good CO_2 sequestration capacity. Among these are species that produce tannins, dyes, elastomers, herbal medicines, food and other products. These trees have contributed to attracting a large number of mammals, birds, insects, bats, amphibians and reptiles back to areas that were once part of their natural habitat (Asorpar Ltda and South Pole Carbon, 2011).

Restoration of soils degraded by gold mining

Over 100 ha of soils degraded by alluvial gold mining have been reforested and restored by the present project.

Opencast gold mining had been executed on the sites of interest (located in the Lower Cauca region of Antioquia Department, Colombia) using tools ranging from bulldozers, excavators and tractors to small tools such as sieves, shovels and picks. Gold miners typically mixed the organic layer with the rest of the soil, in most cases digging down to the bedrock. This mixture was then washed with water under pressure to pass it through a funnel and a canoe-shaped sieve containing quicksilver (mercury) to catch available gold particles. The contaminated and sediment-loaded water then typically flowed into creeks, streams and rivers.

The mining activities left behind gullies, slag heaps (sterile accumulation zones) and lagoons (Figure 14.4). However, even under such degraded conditions life can show its power to regenerate ecosystems seemingly lost, with the appearance of rustic and fast growing species such as downtree, pedro tomín (*Cespedesia macrophylla* Seem.), yagrumo macho (*Schefflera morototoni* (Aubl.) Maguire, Steyerm. & Frodin), yarumos (*Cecropia* sp.), capulin (*Trema micrantha* (L.) Blume) and jacaranda (*Jacaranda copaia* D.Don).

Several steps were followed in the restoration of these sites, beginning with landscaping, followed by subsoil preparation and finally fertilizing and planting.

Landscape

At this stage the objective was to establish a topography as similar as possible to that of the surrounding area. The waste piles left behind by mining activities, consisting of stones, sand, gravel, silt and, in some cases, organic matter deposited on the sides of gullies, were levelled using a bulldozer. This also improved the soil texture and structure (Figure 14.5).

Earlier projects used heavy machinery (a Caterpillar D-6 bulldozer and a CASE 1450B), but more recent projects used lighter and more versatile machinery (a Caterpillar D4 bulldozer) to reduce soil compaction.

Subsoil preparation

The bulldozer was equipped with a 0.7-m-long hydraulic hook to make grooves while levelling the soil. This broke up compacted soil layers and improved soil physical properties. This encourages root development and improves the establishment and growth of the trees and shrubs planted.

Fertilizing and planting

Fertilizers were applied in the grooves in accordance with recommendations derived from the soil analyses. After this, the trees were planted in the grooves.

Results

The post-restoration project area can be characterized as a polyculture dominated by native species. The exotic species *Acacia mangium* and teak were planted on soils where abiotic conditions impeded the growth of other flora, in order to create favourable soil conditions. Only 21 species

Figure 14.4.
Initial state of the restoration site as left by opencast gold-mining activities

Figure 14.5.
The restoration site after levelling of the soil with a bulldozer. The little naturally occurring vegetation visible in the picture was preserved

were found on the project area prior to restoration; currently, over 100 native species are found in the plantations, established through natural and assisted regeneration. This approach has enabled the restoration of ecosystems in historically degraded areas. The plantations are currently nine years old. The project offers a unique opportunity to gain valuable knowledge about management practices and appropriate selection of native tree species for commercial forestry plantations, particularly with respect to seed selection, planting, management and maintenance.

The reforestation and restoration activities of the project increased economic activity through the sustainable nature of practices being implemented, and the creation of employment for local communities. Even people from local communities who previously engaged in mining practices, were encouraged to participate in this project, which provided economic stability and contributed to improving people's living standards.

Today, the area is a positive example of what can be achieved in a short term by taking advantage of voluntary carbon markets. With around 1292 hectares already planted, the project not only generated 198 000 VCUs (verified carbon units, equivalent to 198 000 tonnes of CO_2 fixed, or the annual average emissions of more than 90 000 Colombians), but also delivered climate, community and biodiversity co-benefits recognized by the Climate, Community and Biodiversity Standards. At the time of this project, the trading price of carbon was more than €8/t (Asorpar Ltda and South Pole Carbon, 2011).

Figure 14.6 provides an overview of the volume of wood and amount of biomass produced and CO_2 sequestered by individuals of different tree species between 2002 and 2010 in areas degraded by mining and intensive cattle-raising.

Acacia mangium showed the strongest growth, greatest uniformity in plantations and best adaptation to soils damaged by mining. It is also leguminous, forming a symbiosis with nitrogen-fixing bacteria and producing abundant leaf litter; this in turn adds organic matter to the soil. As such, the species creates a microclimate in which many

Figure 14.6.
Volume of wood and CO2 sequestered per hectare in project areas previously degraded by mining activities (A) and intensive cattle raising (B)

(A)

(B)

other plant and animal species can establish and thrive and further enrich the understorey. For these reasons, *Acacia mangium* was combined with fast-growing native species, some established by means of the enrichment method and others through dispersal of collected seeds.

The non-contaminated lagoons left by gold mining were used for fish farming with species

Figure 14.7.
Current status of restoration site after 96 months. The site is dominated by pedro tomín (*Cespedesia macrophylla*).

such as red-bellied pacu (*Colossoma bidens*), red tilapia (*Oreochromis* sp.), carp (*Cyprinus carpio*) and small mouth (*Prochilodus magdalenae reticulata*). The lagoons are now surrounded by trees, the water is cleaner, there is less evaporation, temperatures are cooler and the aesthetic aspect of the sites has improved.

The approach described has been replicated at other locations. Of the 12 000 hectares previously degraded by gold mining, more than 50 percent have already been recovered.

The success of this approach depends in large part on the direct involvement of communities living in project areas, respecting their culture, and contributing to the improvement of their living environment and standard of living. After all, one of the most valuable results of the project is to generate alternative ways of life, and at the same time create new sources of income. Thanks to the more than 40 000 days of paid work generated by the project, people have been able to improve their health, quality of life and purchasing power. In addition, the project provided people with guidelines about administrative, environmental, social and financial matters. As the ecosystem recovered, natural control systems against the proliferation of malaria vectors were restored, reducing the impact of the disease.

Conclusions

- The basis and key to success in reforestation projects lies in the quality of the seed being used. Therefore, seed trees must be carefully selected and conserved through vegetative propagation in a clone bank and seed orchard. In addition, seeds should be used from areas belonging to the natural distribution of each species.
- Forest remnants occurring in the vicinity of the plantation should be studied to assess natural regeneration, the dominant species, species phenology, associated fauna, the most common pests and diseases, and other basic information necessary for the

development of the restoration project.

- Only healthy seeds of known origin and from selected seed trees should be planted.
- Undesirable individuals in the forest nursery should be discarded.
- All management oriented to the establishment and maintenance of plantations should be performed in the most natural way possible: always remember that we should be working with nature.
- Combine technology with empirical and natural knowledge. The environment should not be modified to any significant extent, as this will usually damage the project outcome.
- Align management and technical guidelines. Operational resources are already available in the region; farmers are generally innate planters.
- The initial establishment practices (weeding, digging, fertilizing, planting) and the origin of the plant material largely determine the success of the plantation.
- After the plantation has been established, proper maintenance activities (weeding and fertilizing) are indispensable for the development of trees planted.
- Weeding contributes directly to the growth of trees and to the control of pests and diseases.
- Take advantage of the land between the tree stands, for example, by means of agroforestry or silvipastoral systems. Such practices result in resources being reinvested in the project and make it more profitable. They also help to control pests and diseases and contribute to making fuller use of fertilizers.
- Encourage local people to be involved in the establishment and management of the plantation, so as to enhance the chances of success and significance of the project. Local communities are the beneficiaries of the project, receiving the vast economic and social benefits from reforestation.
- Care for wells, channels, creeks and rivers. No weeding or cutting of forests surrounding swamps and lagoons should be allowed.
- Protecting forests by means of fences and surveillance by rangers guarantees the longevity of future forests.

Recommendations

- More research should be conducted by government agencies on forest-tree breeding and production. If this is not done, valuable and fast-growing species for potential use in reforestation pilot schemes will disappear.
- Continue to select superior or plus trees at various sites within the natural distribution of species to obtain a broad genetic base of planting material with a wide range of shapes, volumes and physical, chemical and mechanical properties.
- There is a need to collect as much information as possible about the selected plus trees in order to determine their edaphic requirements, production potential, best planting distances, susceptibility to pests and diseases, enrichment methods, behaviour of the species, phenology and germination potential of seeds, nursery propagation techniques and plantation systems.
- Insurance policies covering abiotic and biotic risks in the plantations should be promoted to encourage and secure investment while attracting capital.
- Advantage should be taken of forest plantations and their surroundings to install research programmes funded by the government to encourage scientific work.

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Chapter 15

Species restoration approaches

15.1. Species restoration through dynamic *ex situ* conservation: *Abies nebrodensis* as a model

Fulvio Ducci[27]

Consiglio per la Ricerca e la sperimentazione in Agricoltura, Forestry Research Centre (CRA-SEL), Arezzo, Italy

Abies nebrodensis (Lojac.) Mattei, or Sicilian fir, is an endemic species of Sicily, Italy, growing on the Madonie range at 1700–1900 m above sea level. It is a highly endangered species (Conseil de l'Europe, 1977; Morandini, 1986; Raimondo *et al*., 1992; IUCN, 2007; Thomas, 2011), comprising a single relict population of approximately 30 adult trees spread over an area of 150 hectares (Morandini *et al*., 1994; Virgilio, Schicchi and La Mela Veca, 2000; Figure 15.1). Specific measures are needed to conserve and restore genetic variability of the species in a dynamic way that enables continued evolution of the species, particularly as climate changes.

Although an extreme example, *Abies nebrodensis* can provide useful insights for the conservation and restoration of other species and their genetic variation. The populations of many tree species are fragmented in the southern part of their distribution range, yet these populations may represent valuable sources of genetic information under changing environment. The factors that affect their genetic patterns and viability must be understood and managed in order to develop methods for dynamically preserving such gene pools. Climate change and ecological changes in the Madonie range are rendering the locale unsuitable for the long-term maintenance of the existing gene pool of Sicilian fir, indicating the need to adopt pragmatic strategies to preserve the species. Environmental or practical considerations do not always allow such conservation *in situ*, in natural populations. This article presents options for species restoration and dynamic gene conservation *ex situ*, based on recent genetic studies on *Abies nebrodensis*.

Status of Abies nebrodensis

Abies nebrodensis was more widely distributed in the past in the Madonie range in Sicily, as shown by fossil wood samples of about 9000 years old (Biondi and Raimondo, 1980). The range of the species declined rapidly at the end of the seven-

[27] Coordinator of FAO *Silva mediterranea* WG 4 "Forest Genetic Resources in the Mediterranean Region".

Figure 15.1.
Current natural distribution of *Abies nebrodensis*. The within-population genetic structure increases progressively from the centre towards the peripheral rings in the diversity core zone (orange concentric rings, Vallone Prato). The extinction phase was detected in the Contrada Timpa Rossa area, while the recolonizing phase (blue line) is associated with only two trees near the Vallone della Madonna degli Angeli.

Source: modified after Ducci *et al*. (1999); De Rogatis, 2011, unpublished data.

teenth century, when demand for timber in the neighbouring villages increased at the same time as grazing expanded to higher elevations.

Local botanists have known of the existence of a peculiar population of *Abies* in Sicily for several centuries. Indeed, samples of the Sicilian fir can be found relatively frequently in herbaria dating from the end of the eighteenth century (Raimondo, Giancuzzi and Schicchi, 1990). *Abies nebrodensis* was first described as a separate species in the beginning of the twentieth century (Lojacono Pojero, 1907; Mattei, 1908). Several authors related *A. nebrodensis* to *Abies alba* Mill. (Silver fir), and in particular to the nearby Calabrian *A. alba* provenances, on the basis of morphology (Arena, 1959; Gramuglio, 1960; Morandini, 1969) and using needle anatomical traits (Bottacci, Gellini and Grossoni, 1990). However, *A. nebrodensis* seems to derive from the pre-glacial group of several fir species which are the ancestors of *A. nordmanniana* (Steven) Spach (Caucasian fir), *A. cilicica* (Antoine & Kotschy) Carrière (Cilician fir), *A. cephalonica* Loudon (Greek fir), *A. numidica* de Lannoy ex Carrière (Algerian fir) and *A. alba* in southern Calabria. Genetic distance analysis (anatomical, biochemical and molecular) showed a clear discrimination of *A. nebrodensis* from the other species (Ducci *et al.*, 2004; Camerano *et al.*, 2012). At the same time, this species has preserved some traits typical of more eastern fir species (Ducci, Proietti and Favre, 1999; Ducci *et al.*, 2004) as well as traces of ancient exchanges with geographically nearer species (*A. alba,* and *A. numidica*). In some cases, *A. nebrodensis* showed an allelic pattern completely different from that of neighbouring populations of *A. alba* but closer to that of the oriental species. Results of the genetic analyses were similar for needle morphology and anatomical traits. Together, they suggest that *A. nebrodensis* could be the focal point where *A. alba* traits, *A. cephalonica* traits and *A. numidica* traits converged in the past. For phylogenetic and genetic studies on *Abies nebrodensis*, see Vicario *et al.* (1995), Parducci *et al.* (1996, 2001) and Ducci *et al.* (2004).

Research aimed at conservation of the relict *A. nebrodensis* population started in the 1940s, and initially focused mainly on the biology of the species, monitoring the population and restarting the local forest ecosystem succession. Scientific techniques used in this period included inventories of existing trees, periodic surveys of sexual reproduction, anatomical studies on adaptive traits on needles, seed germination, vigour and phyto-ecological aspects in the area. Those studies supplied clear evidence for the high potential of the population to restore its dynamism, which has since been confirmed by phylogenetic and genetic studies within this population. In the inventories carried out in 1992 (Morandini, Ducci and Menguzzato, 1994) new trees were recorded *in situ*, and a very slow but progressive restarting of the forest ecosystem was documented. Starting from only two or three flowering trees in 1960s, the amount of pollen and cones produced by the population has significantly increased (Arena, 1960; Gramuglio, 1962).

Genetic diversity within the *A. nebrodensis* population was estimated using allozyme markers (11 loci, 32 alleles; Ducci, Proietti and Favre, 1999). The within-population genetic structure was compared with other Mediterranean fir species and another reference set of 16 Italian populations of *A. alba*. These results showed how certain alleles have contributed to differentiating the *A. nebrodensis* population from the reference populations of the other Mediterranean fir species. Moreover, it was discovered that the genetic diversity within the *A. nebrodensis* population, despite there being only 30 remaining adult trees, was similar to the wider and dynamic populations of *A. alba* growing in analogous isolation and progressive drifting situations (i.e. the forests of Gariglione in Calabria and La Verna in Tuscany). Simultaneously, a very high excess of homozygotes was detected in the *A. nebrodensis* population. The microenvironmental diversity *in situ* has allowed the maintenance of very high within-population diversity parameters and relatively good genetic structuring compared with wider ranging and more numerous populations of Mediterranean firs. The species has a well-defined genetic structure that is characterized at topographic level within the 150 ha of the present natural range. Genetic analyses have shown clearly that the closer the ecological conditions are to the species optimum, the higher the genetic variation and the greater the ability of the population to preserve its gene pool (Vendramin, 1997; Ducci, Proietti and Favre, 1999).

On the basis of the genetic analyses, three distinct zones were identified within the *A. nebrodensis* population: the diversity core of the species, one site in recolonizing phase and one site in an extinction phase (Figure 15.1). The comparison of results of enzyme analyses carried out in 1999 (Ducci, Proietti and Favre, 1999) and amplified fragment length polymorphisms (AFLPs) (De Rogatis, unpublished data) confirmed the geographic distribution of the variation of the population *in situ*. The genetic situation of the Sicilian fir within each zone reflects the microenvironmental conditions, which can be classified as favourable, highly favourable and unfavourable for survival and natural regeneration in the three zones.

The relatively good situation of the population from the biological point of view is in contrast with the generally unfavourable site characteristics, which limit the survival possibilities of the present *A. nebrodensis* population. The present refuge is restricted to a mountain top, which precludes natural spread. The ecosystem has been severely affected by human activities, particularly deforestation and grazing by goats. Only the recolonization zone of Vallone della Madonna degli Angeli is really suitable for natural regeneration. Furthermore, climate change is expected to modify the local environmental conditions as climate belts move upward. In the case of Vallone della Madonna degli Angeli (about 1800 m), the trees cannot migrate to a more suitable environment. Indeed, rocky soils do not allow natural regeneration and migration outside the Vallone and the trees at the mountain top have no possibility of migrating to a higher elevation. Other adverse ecological factors include a progressively increasing risk of fires.

Genetic effects add to the constraints imposed by environmental factors on regeneration. Self-pollination reduces genetic variability across the population, and the long distances between the trees encourage selfing, leading to progressive genetic erosion. The few seedlings currently growing in the core zone all originated from a single mother tree (no. 21, Figure 15.1). Thus, the only area where the population might regenerate is characterized by a severe "founder effect," the loss of genetic variation that occurs when a new population is established by a very small number of individuals from a larger population. The same applies to the new tree generations in the zone undergoing recolonization because of the lack of mother trees that produce viable seed. Even if the present adult trees can survive, microenvironmental conditions that adversely affect natural regeneration and survival of seedlings severely threaten the population, except for the small recolonizing zone of Vallone della Madonna degli Angeli. Here, the genetic variation will be low and influenced by the above-mentioned founder effect.

Action needed for species restoration

Previous work has shown the urgent need to reduce the genetic erosion of the important gene pool of *A. nebrodensis,* which contains traits common to the Mediterranean firs. The first step should be to stop the loss of rare alleles and phenotypic traits of possible adaptive significance. Another goal should be to reduce the strong excess of homozygotes, the presence of which implies constraints in maintaining the high polymorphism observed in the population. This will require random mating among all genotypes.

The present genetic situation of *A. nebrodensis in situ* is potentially suitable for re-establishing the dynamics of the species, but the environmental conditions are not conducive to long-term spontaneous survival. When the present generation concludes its biological cycle, the species will be at real risk of extinction.

Ducci, Proietti and Favre (1999) studied the intrapopulation genetic structure of *Abies nebrodensis* to explore the possibilities of conserving the species *in situ*. The aim of their work was to develop a strategy for the re-establishment of the biological dynamics of the existing gene pool, set bases for future spreading of propagation materials of *A. nebrodensis* in the Madonie range and, in view of the effects of climate change, to establish new and dynamic populations in suitable locations. The authors concluded that practically the only way to achieve these goals, and the one involving least risk, would be *ex situ* conservation using seed orchards devoted to increasing mixed mating in the next generations.

There have been some previous attempts to propagate *A. nebrodensis* through seed collection and grafting. However, even if it is possible to obtain large numbers of seedlings under nursery conditions, moving seed from the natural population reduces the number of seeds available to support species survival *in situ* and increases genetic erosion. Moreover, the small *in situ* areas still suitable for reforestation programmes would then be planted with materials mainly from only three or four genotypes that have regularly reproduced *in situ*. This approach would remove any possibility of reducing genetic erosion by introducing reproductive materials with higher levels of diversity. Collection of cones within the Strict Reserve of the Regional Park of Madonie Mountains should be completely forbidden or at least strictly controlled and monitored.

A few grafts of *A. nebrodensis* have been produced and distributed among several European arboreta in the past. Regrettably, the source of the grafts has not been documented and it is possible that they originate from a single specimen (growing within the garden of the Villa of Casale baron in Polizzi Generosa). To our knowledge, the following grafts are still growing:

- Five grafts (unknown original genotype)[28] in the arboreta of Barres and Amance, France, prepared by Dode in 1930 (Morandini,

[28] Technically defined as an ortet.

1969). The grafts produced viable seedlings through open pollination (Fady, personal communication).

- Three grafts (only one mother tree [ortet], identified by genetic analyses) within the garden of Villa Lanza near Gibilmanna (Palermo, Sicily) (Morandini, 1969).
- One graft, probably from the same ortet as above, at the arboretum of Borde Hill (Sussex, United Kingdom) of which, however, no recent news is available.
- In addition, seedlings of *A. nebrodensis* are known from the following locations:
- About 40 seedlings, 10 to 12 years old (provenance Vallone Madonna degli Angeli, unidentified mother trees), were planted near Papiano (province of Arezzo, 43°48′55.008″N, 11°41′46.68″E, 752 m above sea level, southwest aspect).
- Eighty seedlings, of the same age as above, near the State Forest Service nursery in Pieve Santo Stefano (province of Arezzo, 43°39′16.5954″N, 12°3′22.68″E, 700 m above sea level, northwest aspect). They are now monitored annually for pollen and cone production.
- Three 25-year-old trees in the arboretum of Vallombrosa (province of Florence, 43°43′52.392″N, 11°33′18.3594″E, 1000 m above sea level, westerly aspect).

Other grafts were distributed to private arboreta in France (M. Albert Dumas) and in Germany (Herr Klaus Albert Höller).

All these collections can contribute to *ex situ* conservation of *A. nebrodensis*, but they are characterized by the absence of information about mother trees or ortets. Genetic analyses are being carried out to determine the parental origin of these materials, but most of them have to be excluded as seed sources because of possible hybridization with other species or self-pollination. The importance of establishing seed orchards devoted to re-establishing a dynamic genetic structure of the species is clear. The materials should be characterized by greater heterozygosity and diversity than the previously used seedlings. The *ex situ* conservation approach should be integrated with *in situ* conservation approaches.

Species restoration through dynamic *ex situ* conservation

A specific programme for the *ex situ* conservation and restoration of *Abies nebrodensis* was started in 1992 in response to the endangered status of the species and the anticipated effects of climate change. The aim of the programme was also to use *A. nebrodensis* as a model for studying *ex situ* conservation and species restoration methods for small or marginal population gene pools.

To produce reproductive material able to restore the genetic dynamism in *A. nebrodensis*, two orchards (clone archives) were established in 1994 in Tuscany near Arezzo (Pomaio 43°28′39.36″N, 11°57′0.72″E, 690 m above sea level and Caprile, 43°43′34.212″N, 12°7′6.96″E, 910 m above sea level), 1200 km north from the original population (Figure 15.2).

The experiment represents a model strategy to conserve *ex situ* the entire gene pool of a species through massive clonal replication. The orchards were established with grafts from the 27 adult trees of the original population. Materials were grafted onto four-year-old rootstocks of *Abies alba* (provenance Serra S. Bruno, Calabria) in pots. Each mother tree (ortet) is replicated at least five times. In total, about 200 grafts grow in the orchards, and each year some of them are rejuvenated in order to have new clonal copies. The orchards were established according to a complete single-tree random design to improve pollen exchange among genotypes and increase heterozygosity, which in the natural population are constrained by the very low density of adult trees.

The clones in the lower-altitude orchard started to reproduce in 1997, with 80 percent of the clones producing male flowers. Two clones (1 and 22) produced the first cones in spring 2000. In total, 110 seedlings survived in the nursery. In spring 2005, three clones (17, 22 and 29) produced new cones with about 300 g of seeds, of which half were sown and the rest dried and conserved

Figure 15.2.
The natural range of *A. alba* (light green) and of *A. nebrodensis* (orange cross) in Sicily, and the location of the clonal orchards and *ex situ* dynamic conservation test in Tuscany (orange circles)

at –5 °C. About 150 seedlings were obtained. In spring 2007, two clones (1 and 6) produced cones, and in 2010 about 110 new seedlings were obtained from six ortets (1, 6, 13, 17, 18 and 24). Numbers of seedlings produced seem to be low, as a result of the young age of the orchards and the low rates of germination. A cut test showed an average of about 20 percent empty seeds, and germination rates were extremely low, 1–39 percent, very similar to the original population. Nevertheless, each year more maternal genotypes are producing seeds and open-pollinated offspring (Figure 15.3).

The conservation programme was continued in 2007 with the establishment of two permanent experimental plots for dynamic *ex situ* conservation of *A. nebrodensis*. In this model, the gene pool of the original population is introduced and tested within a new ecological context. The effects of different patterns of driving forces (e.g. more light, more drought or more continental climate) are assumed to increase the possibility of conserving different alleles in different combinations.

The *ex situ* conservation plots are located in the northern Apennines (1000 m above sea level, northern aspect) and were established in collaboration with the State Forest Service, Forest Biodiversity Bureau of Pieve S. Stefano (Arezzo). The plots differ in their ecological contexts. One of the plots is under the canopy cover of sweet chestnut (*Castanea sativa* Mill.) and other noble hardwoods at very low tree density. The other plot was planted in an abandoned field area surrounded by a forest consisting of European beech (*Fagus sylvatica* L.), Turkey oak (*Quercus cerris* L.) and hop hornbeam (*Ostrya carpinifolia* Scop.) (Figure 15.4).

The new *ex situ* populations are "open," in that they were established using material from more than one seed collection (i.e. from successive years and from different genotypes). The idea is to plant the offspring produced in the two seed orchards year after year. The establishment of grafted seed orchards allows the replication of the source genotypes several times. This way all the adult trees of the population have been

Figure 15.3.
A grafted *Abies nebrodensis* tree producing cones in one of the Tuscan orchards

rescued from possible random genetic losses, and the homozygote excess will also be reduced efficiently.

Both plots are planted according to a random design in order to create maximum opportunities for outcrossing within the population. Together with the randomized design, having such different microenvironmental contexts contributes to maintaining allelic richness and the variability of genes or gene patterns responsible for adaptation. Moreover, offspring and consequently the gene pool of the population are being introduced in new ecological contexts in order to start new dynamics that may help increase heterozygosity. Nevertheless, the evolutionary forces in these new contexts may be different from those acting *in situ* and consequently the populations may adapt and evolve differently. Monitoring the genetic dynamics in *ex situ* conservation and the genetic variation in offspring is one of the aims of the experiment, in order to have a reference model of the possible genetic implications of such conservation approaches.

Next steps in the *ex situ* conservation programme will be artificial flower induction in the orchards, and the use of genetic markers to monitor the genetic structure of the new generations produced. This will allow the orchards to supply *in situ* nurseries with reproductive materials that have been monitored and checked for their genetic structure.

Experiences gathered during these research activities have allowed local Sicilian bodies to develop strategies for dynamic *in situ* conservation of the relict *A. nebrodensis* population. In the 2000s the Ente Parco delle Madonie, the regional entity responsible of the management of the Regional Park of the Madonie Range, where the Sicilian fir is protected *in situ*, established a strategy for developing *in situ* conservation initiatives. A Life Project on "Conservation *in situ* and *ex situ* of *Abies nebrodensis* (Lojac.) Mattei" was approved and funded in 2000.[29] A second project, "Conserving *Abies nebrodensis* and restoring the bogs of Geraci Siculo," was approved and funded in 2004.[30] These projects focus on preserving the individuals still surviving *in situ*. As in our project, they grafted and established a clonal orchard to preserve the original gene pool *in situ*, with the aim of increasing the species' distribution and starting future reforestation programmes. So, two parallel strategies were developed, one *in situ* and another *ex situ*.

[29] http://www.parcodellemadonie.it/doc/Relazione_progetto_di_recupero_Abies_nebrodensis.pdf

[30] http://www.parcodellemadonie.it/conservazione-di-abies-nebrodensis-e-ripristino-delle-torbiere-di-geraci-siculo.html

Figure 15.4.
Ex situ conservation plots of *Abies nebrodensis* (a) under the cover of chestnut and maple trees and (b) in an open field near a *Fagus sylvatica–Quercus cerris–Ostrya carpinifolia* forest

(a) (b)

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15.2. Restoration and afforestation with *Populus nigra* in Hungary

Sándor Bordács and István Bach

Central Agricultural Office, Hungary

In the 1990s, European black poplar (*Populus nigra* L.) was declared an endangered species in Europe because of widespread planting of hybrid poplar and genetic hybridization and introgression by pollen flow from hybrid poplar clones (mainly *P. × euramericana*).

Since 1997, reforestation or afforestation on protected areas in Hungary has been carried out using only autochthonous species, as mandated by the National Law of Nature Protection. This regulation has increased demand for reproductive material of European black poplar for afforestation on protected sites, mostly in the former willow–poplar riparian forests (*Salicetum albae-fragilis*) near riverbanks. In central Europe the river bank forests are composed of black and white poplar (*P. nigra* and *P. alba* L.), silver willow (*Salix alba* L.), and on less wet sites (higher elevation) of narrow-leaved ash (*Fraxinus angustifolia* Vahl) and pedunculate oak (*Quercus robur* L.).

Figure 15.5.
An old black poplar tree in the Danube valley in Hungary

A national programme was started in Hungary in 1997 to restore riparian forests. As black poplar was a fundamental species for riparian forests and there was no autochthonous reproductive material available, genetic monitoring had to be carried out. First, registered occurrences in the national forest inventory were surveyed. The first survey listed 4390 ha of forest in Hungary, mostly mixed forest stands of *P. nigra*. Thousands of hectares were surveyed to develop a distribution map of existing populations, forest stands, small patches and even solitary trees (Borovics *et al*., 1999). Since 1998, about 4000 mature trees have been selected across the country for the gene conservation and restoration programme. All trees have been characterized morphological descriptors published by the European Forest Genetic Resources Programme (EUFORGEN) to exclude artificial or natural hybrids (*P.* × *euramericana*). Propagation material, usually stem cuttings, was collected from the trees to establish clonal gene banks at six locations in four geographical regions of Hungary. Additionally, DNA of all sampled trees was tested according to the method used by Heinze (1997) to exclude introgressed and hybrid genotypes. Only clones with proven taxonomic status were registered in the national database. All the gene bank clones were tested for cultivation value (e.g. viability, root formation, growing capacity). The most appropriate clones were certified as basic material in the National List of Basic Materials (NLBM) by the national authority, and clonal mixtures (reproductive material of clonal composition) have been produced and marketed. The clonal mixtures have usually been composed of 30–60 clones and are recommended for use in the same region in which the provenances originated. Since 1999, when the first stock plantations were established, stem cuttings have been increasingly produced. Usually, the stem cuttings are used in nurseries where rooted cuttings (height 150–300 cm) are produced.

Some of the selected populations and stands (according to EUFORGEN criteria) have been officially registered as seed stands in the NLBM. As a minimum requirement, the stands must comprise both female and male trees and be at least 1 km from any hybrid poplar plantations or black poplar stands with unknown origin. Since poplar seeds are able to germinate for only one or two days, the female (seed) trees must be felled to collect viable seed. Seedlings are produced in forest nurseries. The seedlings must also be tested for purity of their taxonomic status. In late summer, a local inspector collects leaves from the seedlings for DNA testing. Nursery production is under official control to fulfil the requirements of the certification system for forest reproductive material (FRM). Only DNA-tested seedling lots can be certified and marketed.

The certified cuttings and seedlings are used for forest restoration on protected sites, mainly for riverbank forests, to recreate natural forest vegetation. The reconstruction of forest vegetation has been carried out using reproductive material of two to four autochthonous species planted at

Figure 15.6.
Certified reproductive material of black poplar (*Populus nigra* L.) produced in Hungary from 2000 to 2010

5000–8000 trees/ha. Based on the experiences of local foresters, rooted cuttings can better survive short-term high floods and seedlings can better tolerate long-term lesser flooding with muddy water. Between 2000 and 2010 a total of 120 000 to 1 060 000 black poplar seedlings and 55 000 to 350 000 rooted cuttings were certified as FRM each year and marketed in Hungary (Figure 15.6).

Each year, 30-120 ha of river bank forests are being restored. Black poplar is also being planted in roadside plantations. Reforested areas are under the official control of local inspectors of the state forest service. The inspectors check certificates (supplier's documents) of FRM used for planting and survey the plantations regularly. The basic data and ecological information on the reforestation area must be documented in the forest management plan, including species composition, total area, origin of FRM, and soil type. Management interventions such as nursing and tending may be required. The restoration forests are supposed to be managed less intensively than hybrid poplar plantations, with the aim of establishing forest that is as close to natural as possible.

Information on the black poplar restoration programme and its results have reported and published for the public, especially for local communities and nature conservation organizations. The first positive results from the black poplar restoration programme have stimulated work on gene conservation programmes for other endangered tree species, such as *Sorbus* and *Pyrus* species.

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15.3. Restoration of threatened *Pinus radiata* on Mexico's Guadalupe Island

J. Jesús Vargas-Hernández,[1] Deborah L. Rogers[2] and Valerie Hipkins[3]

[1] Programa Forestal, Colegio de Postgraduados, Montecillo, México
[2] Center for Natural Lands Management, Temecula, California, United States
[3] National Forest Genetics Laboratory, US Forest Service, Placerville, California, United States

Monterey pine (*Pinus radiata* D. Don) is a forest tree species of great economic importance worldwide, although its native range is restricted to the coastal zone of central California and northern Baja California (Figure 15.7). The species is grown in exotic commercial plantations on over 4 million hectares, but in its countries of origin it faces heightened conservation issues; the species has lost over 50 percent of its natural habitat and is threatened by various human-related disturbances. Monterey pine is on the IUCN (International Union of Conservation of Nature) Red List of threatened species, and the FAO Panel of Experts on Forest Gene Resources has identified it as a species with high global, regional and/or national priority for genetic conservation.

The island of Guadalupe in the Pacific Ocean, 250 km off the coast of Baja California, Mexico, hosts one of the five remnant natural populations of Monterey pine. Environmental conditions on the island are harsh, with annual rainfall averaging less than 200 mm. Although dense fogs are common in winter, especially at higher elevations, they are less frequent in summer. The volcanic-origin island has thin, rocky soils with little organic substrate. The Monterey pine population on this island has evolved isolated from the other island and mainland populations, so it has become genetically differentiated from them, showing distinctive morphological and adaptive traits as well as genetic diversity measured with molecular markers. Some authors recognize this population with the varietal name of *P. radiata* var. *binata*. The original pine population once occupied an extensive area on the northern end of the island. However, even though the island has not been permanently inhabited by humans, the pine population shrank dramatically in the last two centuries because of goats that were introduced in the mid-nineteenth century, preventing successful regeneration of the pines. The current population is down to about 220 adult, overmature trees (2001 census), growing isolated or in small patches (Figure 15.8), in an environmental context hostile to recruitment of seedlings. The drastic reduction in population size led to the opinion that this population was headed towards extinction. In 1981, the Guadalupe Island population of Monterey pine was declared "endangered" by the FAO Panel of Experts on Forest Gene Resources largely because of the grazing pressure from introduced goats.

In 2001, a multinational team completed an expedition to Guadalupe Island to make seed collections of Monterey pine for conservation, restoration and research purposes. In addition to collecting seed from individual trees, the team described the status of the pines, evaluated risks and threats, and gathered information on pine ecology to assist in restoration efforts. For example, before the expedition it had been speculated that microsite conditions may have deteriorated to a state that would no longer support seed germination or seedling growth. However, soil and moisture conditions, at least within the canopy and fog-drip zone of living trees, appeared to

Figure 15.7.
Location of natural *Pinus radiata* populations along the west coast of North America. Mainland populations occur at Año Nuevo, Monterey and Cambria. Island populations occur on the Mexican islands of Guadalupe and Cedros

Figure 15.8.
Approximate location of remnant trees of *Pinus radiata* at the northern tip of Guadalupe Island (see also Figures 15.9 and 15.10)

be sufficient to allow at least initial seedling establishment. The discovery of a few small pine seedlings supported the hypothesis that natural recovery may be possible if grazing pressure were reduced or removed. At the same time, a multi-institutional project, led by the Grupo de Ecología y Conservación de Islas A.C. (GECI, a binational non-governmental organization), in collaboration with several institutions from the federal government, was initiated to eradicate the resident goat population. The eradication project started in 2000, when Mexican ranchers from Sonora, assisted by GECI, local fishermen and the Marines stationed on the island, began trapping and removing goats for direct sale and use as breeding stock on the Mexican mainland. To test the response to release from grazing pressure, GECI installed exclusionary fencing in three areas in the pine population in summer 2001. Several years later the potential for natural recruitment of seedlings in the grazing-excluded areas was evident, as was the necessity of complete removal of the goats (Figure 15.9). The goat eradication project accomplished its goal on 2007, with Guadalupe Island being officially declared free of goats.

Environmental conditions conducive to natural recruitment are one important component of the restoration process for this pine population. However, the question remains as to whether levels of genetic diversity in the population are sufficient. Because of the rapid and presumably massive loss of pines, causing fragmentation and drastic reduction in population size, several genetic impacts may have already occurred, including reduction of genetic diversity and increased inbreeding. Thus, to ensure full recovery of the population it was important to evaluate the need for genetic intervention during the restoration process. For instance, if genetic diversity had been drastically reduced in the population, it might be important to reintroduce genetic material from *ex situ* collections. Similarly, if inbreeding were

Figure 15.9.
One of the southernmost isolated, over-mature, *Pinus radiata* trees remaining on Guadalupe Island with the natural recruitment of seedlings moving out of the protection zone under the tree canopy, May 2006

an issue, actions might be necessary to promote cross-pollination and seed dispersal between patches to reduce relatedness among parental trees in the next generation. Based on a spatially representative sample of germplasm from about 35 percent of the current population, genetic diversity, its spatial structure and inbreeding level were analysed using microsatellite markers. The sampling structure allowed comparison of the genetic diversity and inbreeding level in the progeny (seed) to that in the maternal generation (remnant trees). Results showed that, despite the drastic reduction in population size, the level of genetic diversity – both in the parental trees and their open-pollinated progeny – has not been greatly reduced. The data also indicated a minimum of 45 percent cross-pollination in the population. Thus, the genetic information obtained so far does not support the need for genetic intervention to restore this population other than to move seed among resident trees to increase dispersion distance and accelerate connectivity between patches (Figure 15.10). Although the population is still far from restored, the outlook is much more promising now than it was ten years ago.

15.4. A genetic assessment of ecological restoration success in *Banksia attenuata*

Alison Ritchie

School of Plant Biology, University of Western Australia, Australia

Despite the importance of genetic management within ecological restoration for the long-term persistence, functionality and self-sustainability of populations, genetic assessments of the success of ecological restoration remain rare. If a founding population is sourced from a limited genetic pool, genetic bottlenecking and increased inbreeding could potentially occur, reducing the resilience and adaptability of the future population to a changing climate. If the population is established with seeds of non-local provenance, issues such as outbreeding depression could arise and lack of genetic integration with surrounding populations could transpire, influencing the overall success of restoration.

Microsatellite markers were used to assess the genetic success of ecological restoration, asking two main questions: (i) Is sufficient genetic diversity maintained within adult and offspring individuals in restored populations? (ii) Is there genetic connectivity between restored and adjacent natural, undisturbed populations? (Ritchie and Krauss, 2011). Our focal species was the woodland tree *Banksia attenuata* R.Br. (Proteaceae), which is widely distributed across the south-west of Western Australia and is a keystone species used in ecological restoration within this region. The species displays some of the highest outcrossing rates seen among plants (Scott, 1980) and has mixed generalist pollinator mutualisms with nectar-feeding birds (predominantly honeyeaters in the family Meliphagidae), native bees, wasps and introduced honeybees. Both the restored and natural undisturbed populations of *B. attenuata* were located at Gnangara (31°47′09″S, 115°56′32″E), 40 km north-northeast of Perth in Western Australia.

The restoration site was established 14 years ago within a 100-ha leasehold of Rocla Quarry Products (Rocla), one of five sand extraction quarries surrounding the Perth Metropolitan area. The restoration efforts at Rocla satisfied the environmental completion criteria of the time, aimed at restoring species richness, plant density and cover to the conditions prior to sand extraction. Restoration of the site focused on two main research areas: (i) seedling recruitment and plant survival and (ii) plant growth and developmental responses to a reconstructed soil environment. Seed pretreatments to enhance germination (e.g. smoke), greenstock-enabling treatments (e.g. tree guards, antitranspirants) and various soil treatments (e.g. mulching, irrigation, ripping and application of soil stabilizers) were used to increase plant survival. As a result, by the third year plant density was 59 plants/5 m^2 and the number of species was 14/5 m^2, compared with 78 plants/5 m^2 and 8 species/5 m^2, respectively, on undisturbed adjacent woodland plots, surpassing previous restoration figures (Stevens *et al.,* in press). Details of the restoration techniques employed can be found in Rokich and Dixon (2007). The restored population of *B. attenuata* comprised approximately 200 mature adult trees. The natural undisturbed population was located adjacent to the Rocla mine and comprised approximately 350 mature trees within naturally occurring *Banksia* woodland that is up to 300 years old.

Microsatellite analysis revealed that similarly high levels of genetic diversity (heterozygosity and allelic diversity) were maintained within both restored and natural populations, and were high for both adult trees and their offspring

Figure 15.11.
Genetic diversity measures in populations of *Banksia attenuata.* Line indicates heterozygosity

(Figure 15.11). Genetic diversity was also similar to that of a large, naturally occurring reference population (Kings Park). There was very weak population divergence between the restored and natural populations, signifying sourcing of seed from local provenances within the restoration project. Genetic structure was undetectable within the restored population; in contrast the natural population showed significant structuring up to 30 m. This reflects the pattern of natural seedling recruitment, as opposed to man-made mixing and broadcast seeding used in the establishment of the restored population.

We observed complete outcrossing, very low biparental inbreeding and low correlated paternity in both restored and natural populations, indicating similar patterns of contemporary mating at both sites. Paternity analysis revealed extensive pollen dispersal within and among the two populations (Figure 15.12). Pollination events were recorded at distances of 2 m to 324 m between paternal plants, and more than 50 percent of paternity was assigned to sires beyond the local populations (Figure 15.12).

In conclusion, we discovered that the restored population was successfully integrated with the natural population, likely because of the initial sourcing of genetically diverse seed from local genetic provenances. We also observed the delivery of robust pollinator services to the restored population. This study has revealed that, with the application of best practices for seed collection and restoration, the desired genetic goals for successful genetic management of a keystone species within a post-mining restoration site have been achieved.

Figure 15.12.
Network of inferred pollen dispersal events using a paternity maximum likelihood exclusion analysis. Dark grey area is the natural, undisturbed population and lighter grey areas are the restored populations of *Banksia attenuata*. Each circle indicates the position of a sampled tree, with open circles indicating maternal trees that were also sampled for seed.

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Part 4

ANALYSIS

Chapter 16

Analysis of genetic considerations in restoration methods

Riina Jalonen,[1] Evert Thomas,[1] Stephen Cavers,[3] Michele Bozzano,[1] David Boshier,[1,3] Sándor Bordács,[4] Leonardo Gallo,[1,5] Paul Smith[6] and Judy Loo[1]

[1] Bioversity International, Italy
[2] The Centre for Ecology and Hydrology, Natural Environment Research Council, United Kingdom
[3] Department of Plant Sciences, University of Oxford, United Kingdom
[4] Central Agricultural Office, Department of Forest and Biomass Reproductive Material, Hungary
[5] Unidad de Genética Ecológica y Mejoramiento Forestal, INTA Bariloche, Argentina
[6] Seed Conservation Department, Royal Botanic Gardens, Kew, United Kingdom

This chapter discusses genetic aspects of current practice for ecosystem restoration using native tree species. Practical implications for the viability of the restored tree populations are analysed, and options are presented for improving restoration success by applying genetic principles. To this end, the restoration methods and approaches presented in Part 3 were analysed from a genetic perspective, based on the theoretical and practical issues of ecosystem restoration introduced in Part 2. Authors of the methods described in Part 3 were surveyed to collect additional information about the genetic considerations of the restoration methods presented. The survey included questions about species composition, source of propagation materials, and practical details of the design and implementation of each method.[31]

In total, 23 survey responses were received. Most of the respondents carry out applied research, developing and testing a range of restoration approaches and methods that use native species. A rigorous quantitative analysis of the survey results was not undertaken, as the methods and experiences included do not represent a random sample of ecosystem restoration efforts globally. Nevertheless, the responses can be considered indicative of general trends in ecosystem restoration with respect to genetic aspects, and provide useful information to guide the incorporation of genetic considerations in restoration projects.

This chapter includes a brief summary of survey results on identified key areas, followed by a broader literature-based discussion of the issues and recommendations for research and action.

16.1. Appropriate sources of forest reproductive material

Survey results: *Only half of the restoration methods incorporated guidelines or recommendations for the collection of forest reproductive material (FRM). Such recommendations were clearly more common for approaches aimed at conserving or restoring populations of particular tree species than for approaches that focused on restoring*

[31] The survey is available from https://www.bioversityinternational.org/fileadmin/user_upload/SoW_FGR_RestorationSurvey.pdf.

habitats or ecosystems in general. Sourcing FRM locally or from similar ecological conditions was considered ideal for nearly all restoration approaches. For approaches focusing on ecosystem restoration, distance of the source of FRM ranged between a few hundred metres to approximately 100 km from the restoration site, but typically FRM originated from within a few kilometres of the restoration site. Half of the respondents indicated that lack of populations of the target species in the vicinity of the restoration area very often limited the availability of FRM.

Ecosystem restoration efforts commonly source FRM locally. An emphasis on the use of local germplasm is likely linked to the assumption that FRM will be well adapted to the ecological conditions of the restoration site, although the reasoning is not always stated. In fact, the excessive focus on "local" germplasm may obscure the fact that geographical proximity to the restoration site is not necessarily always the best indication of the quality or suitability of FRM. The problem is exacerbated where remaining forests near the restoration area are fragmented, as trees may suffer from inbreeding, low fitness of progeny or other negative consequences of small population size, and may not constitute good seed sources (see Chapter 2; Lowe *et al.*, 2005; Eckert *et al.*, 2010; Vranckx *et al.*, 2012). These conditions can be assumed to be common in most areas where restoration efforts are underway, yet recommendations for the collection of FRM in such situations are generally lacking. Environmental changes may also already affect genetic quality of tree populations as sources of FRM, although the impacts are not well understood. Quality of existing forest patches as sources of FRM must be carefully evaluated in the light of past or ongoing silvicultural management practices and other forms of resource use or disturbance (Lowe *et al.*, 2005; Schaberg *et al.*, 2008). For example, some logging methods may modify the mating system of the residual trees and result in increasingly inbred seeds through selfing or mating between related individuals (Ghazoul, Liston and Boyle, 1998; Obayashi *et al.*, 2002), compromising the population as a quality seed source. In such cases, sourcing FRM from further away, albeit from similar ecological conditions, may be a better option than resorting to nearby fragmented or (genetically) degraded forests or isolated trees.

However, any introduction of genetically distinct FRM, even of native species, holds risks. If FRM is not adapted to the conditions on the restoration site, severe consequences may result, such as low seed germination or mortality of the plants before reproductive age. Alternatively, and probably more typically, lack of or poor adaptation to site conditions may be expressed more gradually, for example through slower growth or lower survival rates. The first generation of trees plays a key role for subsequent natural regeneration on site, and low genetic diversity in this founder population may result in deteriorating genetic health and fitness over following generations (Reed and Frankham, 2003; Rogers and Montalvo, 2004; see also Insight 1: Examples illustrating the importance of genetic considerations in ecosystem restoration). If species being introduced are the same as or closely related to the species remaining on the restoration site but from genetically distinct sources, there is an additional risk of genetic contamination (Ellstrand and Schierenbeck, 2000; Ellstrand, 2003; Rogers and Montalvo, 2004). Gene flow between native resident populations and non-local introduced plants might lead to outbreeding depression. This refers to a situation where, after repeated crosses between local and introduced provenances, hybrid progeny show lower fitness than local progeny because of the breakup of co-adapted gene complexes by recombination. The phenomenon of outbreeding depression is commonly discussed, although there is still little hard evidence of its effects in trees (Frankham *et al.*, 2011). This might be because its effects may emerge only after several generations (Rogers and Montalvo, 2004) or because many tree species have regular long-distance dispersal events, resulting in sufficient gene flow to avoid complete genetic isolation of populations even when they are geographically distant from each other (Ward *et al.*, 2005; Dick *et al.*, 2008). Therefore, outbreeding depression seems most likely to be a risk only where FRM is introduced from provenances very remote or isolated from the local one.

Other than the geographical origin of FRM, genetic considerations seem to receive relatively little attention in restoration efforts, in spite of their ecological and economic importance (in terms of return on investment; Le *et al.*, 2012). Descriptions of restoration methods do not consistently incorporate guidelines for the collection of FRM, for example on the number of trees from which FRM should be collected or the distance between seed trees. Some of the guidelines that do exist may not be adequate; for example, a recommendation to collect seed from (at least) five trees that we came across during this study clearly falls short. Moreover, even when adequate guidelines are available, they may not be followed for practical or other reasons. Contributors to this study indicated, for example, that good parent trees may already have been removed from the landscape or may be difficult to access, and that it is often very difficult to get people to collect FRM from more than one tree per species even when forest areas remain near the restoration site. Adherence to the guidelines may be even more difficult to evaluate and ensure when collection of FRM is outsourced.

Rules of thumb have been developed for how many samples one should collect to capture at least 95 percent of genetic variation (measured as alleles) with the least amount of effort. Such rules relate to many factors, such as breeding system, pollination system, flowering and seed characteristics (Dvorak, Hamrick and Hodge, 1999; Brown and Hardner, 2000). For example, Brown and Hardner (2000) estimated that some 59 unrelated gametes are required to obtain 95 percent of the alleles in a local population, but twice as many gametes are needed if alleles at different loci are represented at equal frequency. Translated into practical terms, this means that in a completely outcrossing species at least 30–60 randomly selected trees should be sampled (Rogers and Montalvo, 2004). When outcrossing species are open-pollinated and many seeds per plant are available (>30), seed should be collected from at least 15 trees. If the number of seeds per plant is restricted, or there is evidence of mating between full siblings, then seed should be collected from more trees. If there is evidence of substantial self-pollination, a minimum sample of 60 trees is recommended (Brown and Hardner, 2000). Smaller samples are very likely to result in genetic erosion, whereas collecting more than the minimum sample size is recommended when the aim is to maintain genetic diversity (Rogers and Montalvo, 2004). In general, a few seeds from many trees is genetically a more efficient sample of the diversity within a population than many seeds from a few trees (Brown and Hardner, 2000). A number of general guidelines already exist and could be adopted for collecting FRM, such as those published by The Australian Network for Plant Conservation Inc. (Vallee *et al.*, 2004), the University of California (Rogers and Montalvo, 2004) and the World Agroforestry Centre[32] (ICRAF; Kindt *et al.*, 2006). Broad-scale guidelines for the collection of FRM are generally widely applicable and need to be better communicated to restoration practitioners. At the same time, it is important to recognize that the extent and distribution of genetic diversity varies widely among tree species (see Chapter 7), meaning there is also a need for more *ad hoc* guidelines.

If properly designed, individual restoration efforts could also contribute to higher-level goals – in particular to the provision of FRM for future restoration efforts, and the conservation of native tree species and their genetic variation. Such outcomes merit further consideration by restoration practitioners and researchers. Restored forests may later become seed sources for further restoration, both in the landscape through natural seed dispersal and through collection of FRM. This aspect should be taken into consideration when planning restoration, especially for rare, endemic

[32] Also see http://www.worldagroforestry.org/resources/databases/tree-seeds-for-farmers for additional manuals and guidelines.

or endangered species for which the availability of appropriate FRM is often already very limited. Maintaining records on the sources of FRM is essential, as it will inform decisions about future collection sites for restoration materials. Such records will also allow lessons to be learned as the restored forests mature and will permit evaluation of the fitness of the populations (Rogers and Montalvo, 2004). The objective of establishing a future seed source provides further impetus for ensuring a sound genetic basis of initial FRM. Efforts should be made to avoid the conventional risks of "seed collection chains" (e.g. Lengkeek, Jaenicke and Dawson, 2004; Pakkad *et al.*, 2008) whereby successive use for seed collection of planted stands with low genetic diversity exacerbates the effects of a narrow genetic base in subsequent populations. Increased use of certification schemes, such as that of the Organisation for Economic Co-operation and Development (OECD) scheme for certification of FRM (OECD, 2011), and the associated guidelines and protocols should be promoted in this respect, because they not only ensure systematic record-keeping but also traceability of germplasm movement.

Examples included in this study illustrate how conservation of genetic variation is often well integrated in restoration efforts that focus on particular species (see, for example, sections 15.1 and 15.2). In such cases, restoration activities are based on analysis of the genetic diversity of a species within the country or across its distribution range in order to identify distinctive populations and optimal seed sources for each restoration site. Such approaches are particularly valuable for conserving and multiplying the genetic resources of adaptively or historically distinct populations and thus conserving adaptive capacity within a species – even for widespread species that are not threatened. However, lack of knowledge about the extent and distribution of genetic variation in all but a few widely planted, mainly commercial, tree species, especially in the tropics, currently constrains adequate planning of targeted species restoration.

On the bright side, it is becoming feasible to conduct genetic analyses for increasing numbers of species as costs decline and new techniques proliferate. As availability of genetic data for a larger number of species increases, it will become possible to design restoration efforts in such a way that they also contribute to conservation of genetic variation in target species. However, it would be unrealistic to assume that genetic data will become available soon for all relevant native species in restoration projects. In the meantime, guidelines about how to safely extrapolate knowledge about the genetic diversity of well-studied species to broader groups of plants with comparable characteristics would be very helpful to guide decision-making processes in restoration projects. Certain characteristics, such as reproduction mode, breeding systems, and means of seed and pollen dispersal (called life-history traits; Hamrick and Godt, 1990, 1996), have been shown to correlate with patterns of genetic diversity. In the absence of direct genetic information, such traits may provide some guidance about the genetic structure of species, especially in species-rich tropical forests (Rogers and Montalvo, 2004; Vranckx *et al.*, 2012). As such, genetic patterns recorded for well-studied species could be generalized to other species with similar life-history traits to formulate recommendations, for example on the collection of FRM for capturing adequate diversity, risks of fragmentation for the quality of natural seed sources or required population densities in mixed stands for groups of species. Restoration efforts and research are recommended to strive to understand and account for differences in patterns of genetic variation between species groups in order to more effectively capture adequate diversity for establishing viable tree populations for a full range of native species. However, care has to be taken not to over-extrapolate observations made in one particular geographical area or species (Duminil *et al.*, 2007). For example, in some cases, genetically highly diverse tree populations identified in single-species studies were found to coincide with areas that also had the highest species and ecosystem diversity (e.g. Gallo *et al.*, 2009), whereas in other cases quite the opposite was observed, with genetic diversity in widespread species not being congruent with species richness (e.g. Taberlet *et al.*, 2012).

16.1.1. Needs for research, policy and action

- Quantify the risks associated with genetic mismatching resulting from the use of narrow or exotic genetic diversity, including long-term studies. Identify the critical thresholds for genetic diversity in restoration material and the key variables for well-matched sources of FRM. Studies should be initiated in systems that are simple in terms of species and structural diversity to facilitate understanding of genetic and ecological interactions.
- Develop and promote decision-support tools for collecting germplasm for restoration that consider the variation in genetic patterns among tree species and ways of predicting it for lesser-known species based on life-history traits of species. Such tools should allow determination of whether remaining populations of a species in the landscape are likely to contain adequate diversity for sourcing good-quality FRM, and how to identify alternative or complementary sources of FRM when necessary.
- Create wider awareness among restoration practitioners about the risks of using FRM with a narrow genetic base. Promote the adoption of national or international certification schemes, standards and guidelines for collecting FRM and documenting its origin.
- Promote awareness of the potential of individual restoration projects to contribute to species conservation and serve as future seed sources, especially for rare, endemic and endangered tree species. Develop approaches and tools for planning, coordination and communication of restoration activities that support these objectives.

16.2. Species selection and availability

Survey results: *Lack of FRM of native tree species was the most common constraint to the wider application of the various restoration methods. Availability of FRM was limited, above all, by lack of knowledge of species biology (e.g. phenology or propagation methods) and lack of populations of the target species in the vicinity of the restoration area. Availability of FRM and ease of propagation and cultivation were the most important reasons for the choice of species after the successional characteristics of the species, and were considered more decisive than, for example, functional characteristics or conservation status of the species. Most respondents implied that FRM was collected and nursery seedlings were raised as part of the restoration effort. One out of four respondents reported that exotic species were regularly used. The most common reasons for the use of exotic species were their functional characteristics or product preferences.*

In spite of the growing recognition that native species are best for ecosystem restoration, their wider use often seems to be constrained by the lack of knowledge about their biology, such as phenology or propagation techniques, and difficulties in sourcing FRM. Limitations in knowledge may be so severe that they compromise the optimal selection of species for restoration and result in including exotic species because native alternatives are not known. Yet the Leguminosae family, for example, is known to comprise more than 23 000 species,[33] many of which are nitrogen-fixing, which implies wide-ranging possibilities of using native legume species for site amelioration in virtually any area targeted for restoration.

While large gaps remain in knowledge about the biology and ecology of most native tree species, especially in the tropics, it is noteworthy that a considerable amount of useful information about many species has already been collected over the years in various studies and field

[33] http://www.theplantlist.org

visits. Much of this information is hidden in grey literature, and often written in language that is inaccessible to those working directly in the field (Boshier *et al.*, 2009). Similarly, several contributors to this study indicated that when they began to develop their respective restoration methods, there was a preference for species that were commonly available and for which some information already existed. As knowledge on (other) native species in the area improved, it became possible to select species more in accordance with their functional characteristics, competitive abilities or other desirable properties. Existing local and traditional knowledge about species, their propagation and management can be an important information source and should be better documented and integrated in restoration efforts (Douterlungne *et al.*, 2010).

The limited number of native species for which information is available from long-term studies and adaptive management, as well as the difficulty of access to that information, forces many restoration practitioners to compromise species selection or conduct their own *ad hoc* experiments. Enhancing the public availability of information on species with potential in restoration efforts would considerably benefit the community of restoration practitioners and researchers. To have the greatest impact, information should be made freely accessible and easily searchable. Translation into non-specialist or local languages should be considered, at least for the most important species in each area. Some publicly available databases and tools with information relevant to restoration already exist. The Agroforestree database[34] and the Useful Tree Species for Africa,[35] both by the World Agroforestry Centre, include information on propagation and distribution of hundreds of tree species. The Tropical Restoration Information Clearinghouse (Environmental Leadership and Training Initiative, Yale University)[36] has developed annotated literature lists relevant to restoration, including grey literature, as well as information on dozens of restoration projects. The Seed Information Database[37] of the Royal Botanic Gardens, Kew, includes information on optimal germination protocols and other traits such as seed storage behaviour on more than 11 000 tree and shrub species (July 2012), and the UK Germination Toolbox,[38] also from Kew, provides detailed information on germination for native species in United Kingdom.

Exotic species are recognized by many restoration practitioners as important under certain circumstances (Newton, 2011; Alexander *et al.*, 2011a; Lamb, 2012). Many exotic species are known to grow well under harsh conditions and to improve site conditions, for example through biological nitrogen fixation. Therefore, they are often included as nurse crops in the choice of initial species to ameliorate the microenvironment on very degraded sites and to facilitate the later introduction of native species that may be less tolerant (see, for example, sections 13.1 and 14.6). In particular, late-successional species may need the protection of nurse plants against drought, direct solar radiation or drying winds, and may be unable to establish on sites where soil structure, chemistry or hydrology differ considerably from natural forest ecosystems. In some cases, planting a non-invasive, non-persistent exotic species may be a more ecologically compatible solution than choosing an ill-adapted population as a source of a native species that may genetically contaminate resident populations (Rogers and Montalvo, 2004). Exotic species may later be intentionally removed (e.g. before they reach reproductive age), or may be outcompeted by developing vegetation once the native vegetation is well established.

In restoration or rehabilitation projects that include production objectives, one frequent mo-

[34] http://www.worldagroforestry.org/resources/databases/agroforestree

[35] www.worldagroforestrycentre.org/our_products/databases/useful-tree-species-africa

[36] http://reforestation.elti.org/

[37] http://data.kew.org/sid/

[38] http://www.kew.org/science-research-data/databases-publications/uk-germination-tool-box/

tivation for using exotic species is preference for specific products, for example valuable timber or fruits. Common perceptions of superior growth or functional characteristics of exotics compared with native species are at least in part related to better knowledge of some traditional socioeconomically important species than of native species. However, according to the contributors to this study, the use of exotic species was commonly motivated by their specific products or services, and seldom only by a lack of information on or availability of native species. When native species were highly valued in restoration efforts, lack of information or FRM on those species may have resulted in their use but in lower numbers, or in use of native species less suited to the site conditions or restoration objectives, rather than resorting to exotic species.

Introduction of exotics in restoration projects should be carefully planned and based on knowledge of the species and their characteristics. This is particularly important in the light of climate change, since changes in habitat conditions may alter species interactions, for example worsening problems related to invasiveness (Hulme, 2012; Lamb, 2012). Research is needed to better understand the ecological and socioeconomic trade-offs between exotics and native species in a variety of contexts, and more specifically the factors that currently limit the use of native species, including lack of knowledge on propagation methods, availability of FRM and limits imposed by people's prejudices.

By far the most common planting material in restoration projects seems to be nursery-grown seedlings, and the possibility of using optimal species combinations and FRM that are both adapted to site conditions and genetically diverse appears often limited by what is available in nurseries. Nursery operations range from very small-scale roadside enterprises to large, efficient, commercial suppliers, and may be commercially driven or directly linked to projects, local communities, institutions or the state. In commercial nurseries the range of available species, particularly native species, is generally low, largely because the production of species is often guided by general demand, supply and ease of propagation **(see** Insight 6: Seed availability: a case study). Nurseries, private or public, have limited resources and logically try to minimize costs associated with collecting FRM of species that are not normally produced or are from remote areas. Storing and growing FRM with a wide genetic base for a large number of non-commercial species can be extremely expensive. As it can take several years from collection until the material is ready for transplanting, nursery operations need to know what future demand will be, or to predict future demand, which involves considerable risk.

Often, little information is available on the origin of germplasm in commercial nurseries and how it was collected (see Chapter 3 and Chapter 9).

Uncertainty about the identity of FRM may increase the risk of genetic mismatching, as the material may represent too narrow a genetic base or be genetically too distant, increasing the risks of maladaption to the target site and genetic contamination (Rogers and Montalvo, 2004). Where data are available, collection practice often targets the "lowest hanging fruit" of easily accessed trees, which may result in genetic bottlenecks (Lengkeek, Jaenicke and Dawson, 2005; see also Lee, 2000; Hai *et al.*, 2008) and associated reduced fitness in planted trees.

Any restoration project that uses nursery seedlings should, where possible, include a nursery strategy from the outset and integrate the costs and time required for the development of FRM. This would help to avoid dependency on commonly available FRM, allow collection standards on genetic diversity to be met and provide sufficient time for propagating the germplasm using methods tailored to the project. The selection of species can then be better guided by analysis of the natural vegetation and the current and future habitat conditions rather than being subject to the vagaries and practicalities of supply. Enhancing the establishment and use of community nurseries could contribute both to the use of an increased number of local tree species and to an improved sense of ownership of restoration activities among local people. Many rural people have a deep knowledge about their environment, including habitat and growth requirements, phenology and usefulness of locally available species. Current initiatives to develop and test community nursery guidelines and protocols through participatory research with local practitioners should be strengthened and intensified to ensure that FRM produced in local nurseries is of good quality, appropriate to the target site and originates from a sufficient number of genetically diverse parent trees. Certification systems such as the OECD scheme for certification of FRM (OECD, 2011) exist to ensure the quality of FRM and reduce the risk of its uncertain origin, and their use should be promoted in ecosystem restoration.

In addition to improving nursery strategies, considering a wider selection of different restoration approaches and types of FRM could help to include a more diverse set of species in restoration efforts. For example, seed banks store large amounts of seed of many species and sources, and many of them also supply seed for restoration or afforestation purposes. Some larger restoration projects have established their own seed banks, to compile and securely store diverse germplasm until required (see Insight 7: The role of seed banks in habitat restoration).

Creating demand for good-quality germplasm of native tree species through political commitments, supportive and regulatory frameworks is necessary for effecting large-scale changes in the production and supply chains of FRM for restoration. Such frameworks should explicitly address species and germplasm selection and the role of native tree species in ecosystem restoration. For example, development of zoning systems for sourcing FRM and mechanisms for their implementation could result in more consistent use of appropriate germplasm in restoration projects. When developing such supportive and regulatory frameworks, appropriate financing mechanisms should also be identified. For example, in many countries large-scale afforestation projects receive subsidies from the government. It would be helpful to analyse the implications of creating or extending such subsidies to ecosystem restoration and conditioning them to the use of adequate germplasm.

16.2.1. Needs for research, policy and action

- Conduct applied research to understand the potential of native species to achieve various restoration objectives in assorted states of site degradation and ecological and socioeconomic contexts. Analyse the ecological and socioeconomic trade-offs related to the use of exotic versus native species, and the factors that currently constrain wider use of native species. Develop knowledge-based decision-support tools for identifying the conditions under which the use of exotic species in ecosystem restoration can be considered beneficial and justified, or risky and best avoided.
- Improve access to information that is relevant for the restoration community, particularly data on the biology and ecology of native species. Encourage restoration researchers and practitioners to share information and contribute results to publicly available databases, and develop new decision-support tools for facilitating the selection of species and restoration methods. Ensure access to information by local restoration practitioners, farmers and other stakeholders by developing and promoting appropriate communication technologies and products and provision of information in locally relevant languages that uses easily understandable terminology and accessible formats.
- Raise awareness among restoration practitioners of the need for early planning of appropriate and adequate germplasm supplies of desired species, including the associated time and costs. Envisage best ways to embed collection of FRM and nursery production in projects from the outset. Improve documentation of

collection and propagation of FRM as well as communication channels, cooperation and feedback loops between seed suppliers, nurseries and restoration projects.

- Analyse the needs and options for support and regulatory frameworks tailored to the restoration of forested ecosystems and production and supply of FRM. Such frameworks should explicitly address the role and use of native species and the minimum set of genetic considerations that should be taken into account, the role and knowledge of local communities and other stakeholders, and capacity strengthening of local nurseries and seed companies, as appropriate. Develop and implement the frameworks based on needs analysis, ensuring adequate financing for the activities.

16.3. Choice of restoration and propagation methods

Survey results: *Nursery seedlings were by far the most common FRM used across the range of restoration methods, followed by wildings and seeds. With few exceptions, the respondents indicated that nursery seedlings were very often used as FRM in the restoration method they presented. Wildings were also used as FRM in most of the cases, and in half of the cases direct seeding was sometimes applied. Although vegetative propagation was mentioned it was not often used. There was a reliance on natural regeneration when constraints to regeneration (e.g. excessive grazing) were solved.*

Nursery seedlings are very commonly used in restoration efforts, and may currently be the most feasible propagation method for many species. However, if direct seeding or vegetative propagation can be used, these techniques are less expensive and generally require less labour and care than production and planting of nursery seedlings (see Chapter 8). The nursery phase generally seems to receive much attention in the production of seedlings, while crucial aspects of identifying the preferred origin or provenance of germplasm, collection methods and tending of seedlings on site after planting are often overlooked. Selection pressures on seedlings in nurseries are quite different from those in a degraded forest. Many more seedlings can reach older life stages in nurseries than would in the forest, where selective pressures are often more severe. Selection in the nursery that mimics natural selection in degraded ecosystems can filter out inbred individuals and produce healthy, vigorous seedlings. On the other hand, some of the selection exercised at nurseries may unnecessarily (and unintentionally) narrow down or cause directional changes in the genetic diversity among seedlings (Gillet, Gömöry and Paule, 2005). It is not well understood to what extent characteristics such as slow shoot growth at seedling stage relate to genetic diversity and viability, and what the genetic consequences may be of systematic removal of certain phenotypes from the nursery seedling pool. Increasing the genetic diversity of planting stock used in nurseries can help ensure that seedlings are fit to survive and establish in the degraded ecosystem in sufficient numbers. Special care should be given to avoid unintentional selection of traits during harvest of seed for propagation. For example, seed dormancy and seed shattering, which can be important adaptive traits in plants, are often selected against and lost unintentionally under standard seed harvesting and propagation practices (Cai and Morishima, 2002). Growth rate and timing of flowering and fruiting are other traits that are subject to unintentional selection. Harvesting seed in a narrow time window can reduce genetic variation in terms of timing of flowering if there is wide genetically controlled variation in flowering and maturation of seed in the parental population. Harvesting seed towards the beginning or end of seed maturity may similarly result in genetic shifts in the trait (Rogers and Montalvo, 2004).

Restoration researchers consistently highlight the need for careful assessment of site conditions and factors that restrict regeneration before choosing an appropriate restoration method and set of species (e.g. FORRU, 2006; Holl and Aide, 2011; section 14.1). In sites with low to intermediate levels of degradation, where soils are largely intact and there are sufficient genetic sources for the next generation (e.g. parent plants or soil seed bank), natural regeneration may be the best choice (Chazdon, 2008). If natural regeneration potential is good, it may suffice to tend natural seedlings already occurring on the site and to control suppressing factors such as competing vegetation or grazing by animals (see sections 12.3, 12.4, 14.1, and 15.3). Seedling recruitment can be promoted by planting fast-growing species to attract natural seed dispersers to the site and to provide shelter for other species that require some degree of shade (FORRU, 2006; section 12.2). Research has shown the effectiveness of this method in recruiting a large number of seedlings and species to the site (Sinhaseni, 2008). In such conditions, investing in large-scale production of nursery seedlings and planting may not be an efficient use of resources. Natural regeneration bypasses the risks associated with introducing FRM and hence promotes maintenance of genetic integrity (Rogers and Montalvo, 2004). Additionally, this approach constantly allows the best-adapted seedlings to be recruited and hence builds in some capability to cope with changing environmental conditions. However, natural regeneration may be susceptible to genetic bottlenecks where seed trees are few.

Natural seed sources, whether on site or from nearby forest areas, must be sufficiently large and diverse to provide a sound genetic basis for the restored tree populations. Seed produced by trees in small fragmented forest patches in a non-forest landscape matrix may be of poor quality because of inbreeding and genetic drift (Lowe *et al.*, 2005; Eckert *et al.*, 2010; but see Chapter 2). Much depends on the pollination and mating system of the tree species in question. For example, the genetic consequences of fragmentation may differ between wind-, animal- or insect-pollinated species and between self-incompatible and self-compatible species (Vranckx *et al.*, 2012; Chapter 4). So far, few studies have examined the genetic composition of tree populations restored through natural or artificial recruitment processes (Pakkad *et al.*, 2004, 2008; Burgarella *et al.*, 2007; Navascues and Emerson, 2007; Liu *et al.*, 2008; Broadhurst, 2011; Ritchie and Krauss, 2012; section 15.4). Research is needed on this topic for landscapes in different states of degradation and for different species to provide guidance for restoring viable, genetically diverse populations through natural regeneration and seed dispersal.

In sites where diverse natural seed sources are lacking, or seed sources suffer from genetic erosion, introducing additional FRM (and genetic diversity) may either be advantageous (Rogers and Montalvo, 2004) or simply the only solution, at least in the short term. In such areas, more research is needed on alternatives to raising nursery seedlings, as well as on using a mixture of different FRM (Chapter 8). Little is known about the most suitable propagation methods for the majority of tropical tree species. While it is estimated that 80–90 percent of all plant species produce orthodox seed, the proportion is clearly lower among tree species in the humid tropics, where only 50 percent of plant species may have orthodox seed (Tweddle *et al.*, 2003; Kettle, 2012) and are thus more difficult to propagate from seed. Future research should seek to determine which types of species are suitable for restoration through direct seeding under what conditions (FORRU, 2006: 62) and how the effectiveness of direct seeding can be improved. Seed banks usually carry out research for developing optimal germination protocols, and may be both valuable sources of information and excellent research partners on species propagation for restoration purposes (Hardwick *et al.*, 2011; see Insight 7: The role of seed banks in habitat restoration).

Vegetative material can be a valid alternative or complement to seedlings for species that are easily propagated vegetatively, especially when FRM is otherwise scarce or lacking (Chapter 8), as may be the case for rare and endangered species. However, suitability for vegetative propagation, for example, the rooting ability of cuttings, may vary between genotypes. There does not appear to be any evidence that a species' ability to reproduce vegetatively is associated with lower levels of genetic diversity, and clonal plants seem to display intermediate levels of diversity (in the overall range of genetic diversity for plant species) (Ellstrand and Roose, 1987; Rogers and Montalvo, 2004). However, at population level, and especially for planted material such as living fencerows, there is evidence of reduced genetic diversity. It is important to ensure that the source material for vegetative propagation is of adequate genetic diversity to avoid inbreeding depression in subsequent generations and ensure future viability and resilience of the stand (e.g. reduce disease susceptibility).

Ultimately, FRM must be chosen to match the environmental conditions, including the level of degradation of the restoration site. Germination and initial establishment of seedlings are the most sensitive phases of the whole development of the restored forest ecosystem. In some cases, changing climatic conditions and moderate to severe site degradation could modify the environmental conditions of a restoration site in such a way that only nursery-grown seedlings or saplings that have already survived through some of the early critical phases can ensure regeneration.

16.3.1. Needs for research, policy and action

- Carry out research to develop and test suitable restoration and propagation methods and decision-making tools for a variety of native tree species and states of site degradation. Research should include analysing the genetic composition of tree populations restored through different approaches and comparisons with existing tree populations in the surrounding landscape or in similar conditions further away to help ensure the genetic integrity of the restored plant communities. Phased analysis to track the development of genetic diversity at restored sites over multiple generations would help to refine guidance for good nursery practice, for example, by assessing the long-term fitness consequences for different species of using seedlings produced under the relaxed selective environment of the nursery.
- Create awareness of the importance of carefully evaluating site conditions as a basis for choosing the restoration approach that best addresses the causes of degradation and the types of FRM most likely to ensure successful establishment of viable tree populations and most efficient in terms of use of resources.

16.4. Restoring species associations

Survey results: *Only a third of the respondents indicated that the restoration methods they had used deliberately considered restoration of species associations or symbiotic relationships.*

Most restoration efforts appear to focus explicitly on restoration of the tree component in forest ecosystems. Such focus may be based on the general perception that trees are commonly foundation species in the ecosystems where they occur, facilitating the occurrence and evolution of other less prominent organisms (Lamit *et al.*, 2011). However, this overlooks the fact that during their growth and development trees themselves interact with and depend on many other species, including pollinators, seed dispersers, herbivores and symbiotic organisms such as mycorrhizal fungi or nitrogen-fixing bacteria. There is increasing awareness that genetic variation in one species affects that in another species, resulting in complex co-evolutionary processes within entire ecosystems (community genetics; Whitham *et al.*, 2003, 2006). In forest ecosystems such relations may arise, for example, when bird or mycorrhizal fungal species preferentially associate with particular genotypes of a tree species. In some cases, species and genotype relationships may have significant impacts on successful establishment of a population (Ingleby *et al.*, 2007) or may ameliorate negative impacts of abiotic or biotic stresses such as herbivory (Jactel and Brockerhoff,

Box 16.1.
Mycorrhizal and rhizobial symbioses

Mycorrhizal and rhizobial symbioses are among the most obvious beneficial associations to consider for successful restoration. For example, mycorrhizal symbiosis is known to improve the vitality of plants under various stresses (Van Tichelen, Colpaert and Vangronsveld, 2002; Domínguez-Núñez *et al.*, 2006), and growth capacity of various tree species planted in truffle plantations in Central Europe was correlated with mycorrhizal colonization (Bratek, 2008). Efficiency of the symbiotic associations is affected by the identity of both the host plants and microbial symbionts. Nutrient acquisition or nitrogen fixation may vary significantly according to the plant–microbe combinations (e.g. Lesueur *et al.*, 2001). Moreover, there is increasing evidence that mycorrhizal symbioses are capable of altering competitive relationships between plants and consequently composition of entire plant communities (van der Heijden *et al.*, 1998; Hart, Reader and Klironomos, 2003). Although natural inocula of mycorrhizal fungi or rhizobia can usually be found in soils on restoration sites, their populations may have changed because of changes in environmental conditions. Mycorrhizal fungal species found in closed forests may differ from those in areas under anthropogenic influence (Öpik *et al.*, 2006). Inoculation of propagules in the nursery with appropriate mycorrhizal fungi or rhizobia or by seed treatment can facilitate and accelerate seedling establishment by increasing the availability of nutrients and water. Importantly, it can assist seedlings in the crucial early years of restoration, for example in overcoming competition with undergrowth or in tolerating the more extreme weather conditions and drought found in degraded open areas. There is clear evidence that inoculation can increase success rates in plantings (Lesueur *et al.*, 2001; Ingleby *et al.*, 2007). Mycorrhizal colonization can be facilitated by the choice of potting medium and nursery practices, such as establishing forest nurseries and storing seedlings near the ground where they can be more easily infected by native soil fungi (Nandakwang *et al.*, 2008).

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2007). Overall, higher species and genetic diversity improves the stability and resilience of entire ecosystems and ecosystem recovery to climate extremes, which is of increasing importance under environmental change (Gregorius, 1996; Elmqvist *et al.*, 2003; Muller-Starck, Ziehe and Schubert, 2005; Reusch *et al.*, 2005; Paquette and Messier, 2010; Thompson *et al.*, 2010; Alexander *et al.*, 2011a; Gerber, 2011; Isbell *et al.*, 2011; Sgro, Lowe and Hoffmann, 2011), although the relationship between diversity and associated species can be complex (Castagneyrol *et al.*, 2012).

The genetic diversity of FRM from genetically distant sources might have consequences for the species associations that they are used to restore, in some cases leading to cascading effects throughout the ecological community. For example, when plant populations are introduced with earlier or later flowering and seed-setting times than the resident populations, this might have consequences for associated animal species such as pollinators or seed dispersers (Rogers and Montalvo, 2004). Analysis of the dynamics of the main energy chains within the ecosystem can help to identify the interdependencies of species and traits.

Restoration projects should, as far as possible, create appropriate conditions to foster restoration of the interactions and associations between species and genotypes. This should both improve success rates for restoration and promote the biodiversity benefits of restoration projects. Most restoration methods rely at least partly on natural recruitment of seedlings to the restoration site, and their success, therefore, crucially depends on the restoration of pollination and seed dispersal within the ecosystem and landscape. Enhancing diverse species associations also promotes optimal use of growth resources at restoration sites. Using pioneer species that form effective symbiotic relationships with mycorrhizal fungi, nitrogen-fixing bacteria or both could contribute to improving site conditions, provide inocula to assist establishment of other species and, as such, help ensure successful restoration. Additionally, other fungi and bacteria that are not necessarily symbiotic can be used in inocula to trigger bioactivation and recolonization of soil life, and to promote decomposition of organic material, thus increasing nutrient availability. This approach has been successfully applied on highly degraded soils, such as gold mine spoils (section 14.6).

Analysis of site conditions as a basis for identifying appropriate restoration methods should include the associated species present or required in the system (e.g. symbiotic or herbivorous species or species that compete for habitat), their life cycles and probable effects on restoration processes. This also requires study of the approaches required for promoting their establishment and survival and restoring and managing their functions. It is noteworthy that exotics can also contribute to species associations, although there is a lot still to learn about the implications and risks related to the introduction of exotics (weeds, herbivores, mycorrhizal and other fungi, bacteria and insects) in specific systems. As a general rule, it is wise to avoid introducing FRM of uncertain origin and on which information is scarce. Lastly, it would be particularly valuable to consider interactions of multiple species at the landscape scale, bearing in mind the potential costs and benefits to other land uses, including other forest types (Carnus *et al.*, 2006).

16.4.1. Needs for research, policy and action

- Analyse the importance and strength of relationships among foundation species, associated organisms and their genotypes and the implications of the relationships for successful establishment of diverse and viable tree populations. Identify success factors and develop practical approaches and guidelines for restoring species associations using different restoration methods and in different ecosystem and landscape contexts. Develop and test models for predicting the likely benefits of restoration to plant-community relationships, biodiversity conservation, and ecosystem function and resilience.

- Raise awareness of the importance of species associations for the successful restoration of ecosystem functions and promote the consideration of species associations in the planning and design of restoration projects.

16.5. Integrating restoration initiatives in human landscape mosaics

Survey results. *Area of application varied widely within and among methods, but the most typical size reported for restoration actions was about 10 ha. Two out of three respondents indicated that landscape connectivity needs to be considered when applying their restoration method. Landscape considerations were most commonly associated with seed dispersal distances from surrounding forests to the restoration site. The majority of respondents considered carbon sequestration and restoration of habitats for flora and fauna as the most common benefits expected from restored forests, while production of timber, fodder or fuelwood were considered important by only half of the respondents. Half of respondents reported that the restoration methods they used could in some cases be applied to agroforestry or other land-use types, integrating livelihood aspects.*

Very little is known about minimum viable population or effective breeding unit sizes of tree species, especially of those that occur in species-rich tropical forests where population densities are often low. However, from theory, it has been suggested that at least 50 unrelated reproductive trees are needed to form a viable population and avoid inbreeding (Brown and Hardner, 2000; Frankham, Ballou and Briscoe, 2002). Estimates of the typical area of application of restoration methods suggest that many sites for ecosystem restoration may be too small to sustain viable populations of tree species on their own. Therefore, it is important to design restoration projects in a way that connects them to existing tree populations in the landscape or to other restoration efforts. Ensuring ecologically and genetically effective connectivity requires that mating systems, pollen- and seed-dispersal distances or landscape permeability to gene flow are taken into account from the outset of restoration projects. For tree stands to contribute to gene flow it is important to make sure that FRM is genetically matched to the remaining (fragmented) populations of the same species (Rogers and Montalvo, 2004). Hence, if FRM is obtained from outside the target area, the risk of hybridization with existing populations needs to be considered (Chapter 2 and Chapter 6).

Species pollinated by generalist pollinators are generally more readily connected within a landscape than species with specialized or low-energy pollinators (Vranckx *et al.*, 2012). It is important to ensure favourable conditions for pollinator survival and mobility, especially for the latter type of pollinators. Connectivity and gene flow are important for both self-compatible and self-incompatible species: lack of cross-pollination can result in increased selfing and inbreeding depression in the former and in reduced seed set in the latter. Site conditions should also be attractive to seed dispersers that promote natural dispersal and recruitment (Markl *et al.*, 2012). Although in many cases knowledge of species biology and ecology may not be readily available, it is important to learn as much as possible before initiating a restoration project. Interaction with indigenous and other local human communities can be very useful and rewarding as they often hold extensive knowledge of the ecology of their local plants and associated fauna. When planning restoration activities, it may in some cases also be useful to use historically reconstructed reference landscapes and ecosystems based on historical ecology techniques (Egan and Howell, 2005). This allows restoration trajectories to be anchored in historical time (Chapter 10).

In many tropical areas, the success of restoration practices depends on the engagement of local communities (Newton, 2011), not as a workforce but as true participants and direct beneficiaries of restoration projects. People will invest more and care more for species and systems that correspond to their own needs and values. It must be recognized that in some cases local people may prefer species for production purposes rather than ecosystem restoration, may not value non-timber forest products as highly as is often assumed, or may prefer exotic species for their marketability, ease of production or other characteristics. It is noteworthy that half of the respondents of the survey conducted for the preparation of this study did not consider the restored ecosystems as potentially important sources of timber, fuelwood or fodder, all of which are products that may often be of particularly high importance for local livelihoods. This may be indicative of typical ecology-oriented objectives in restoration research, whereas restoration practitioners may focus more on the utilitarian value of the restored ecosystems. Nevertheless, there is increasing recognition of the potential of managed ecosystems to provide important ecosystem goods and services, including carbon sequestration, nutrient and water cycling and biodiversity conservation (Thompson *et al.*, 2010). One contributor to this thematic study pointed out that, in many parts of the world, local people have long-standing experience with production systems that incorporate trees and agricultural plants and in many cases are undoubtedly more knowledgeable about the species and appropriate propagation and planting methods than researchers. He noted that what may often be missing is the promotion of the concept that such production systems also have high potential for ecosystem restoration and conservation (see Chapter 4).

To improve the feasibility and the socioeconomic value of ecosystem restoration projects, efforts are needed to better understand and incorporate local people's preferences for species, land uses and management options, and to identify how outside interests can contribute to or interfere with local objectives. Participation of local authorities is important for continuity and coherence of restoration projects at the landscape scale. In addition, education and training curricula need to be broadened to raise awareness among conservation biologists and rural development practitioners about the potential role of on-farm conservation of biodiversity.

Overall, optimal allocation of restoration efforts at the landscape level requires close collaboration and coordination among the various landowners and users (see Chapter 11). Research is needed on the best means to ensure that individual restoration projects add value to the landscape in terms of connectivity between populations and habitats as well as complementarity of land uses and livelihood strategies of local people. Examples of potentially useful approaches include the analysis of the viability of tree populations in providing products and services in diverse land-use mosaics; development of distribution maps and analysis of necessary landscape linkages (e.g. stepping stones, corridors or favourable land-use matrices) and gaps for target species; development of species-specific action plans for conservation through restoration; and regional-scale habitat corridor initiatives. Finally, landscape restoration depends on national public policies and politics. Decisions leading to large-scale forest conversion are taken by national and state governments, and these bodies are also able to take decisions that will reverse the trend. In spite of recent initiatives, landscape-scale restoration efforts are still few and the area of restored forested ecosystems remains small in comparison with the 13 million hectares of forest converted to other land uses each year (FAO, 2006).

16.5.1. Needs for research, policy and action

- Consistently plan restoration efforts at a landscape scale and seek to integrate them into the surrounding land-use matrix or existing networks of habitat corridors. The presence of existing tree populations of target species needs to be explicitly taken into account to facilitate establishment and maintenance of viable tree populations. Develop and promote tools and opportunities for learning, coordination, communication and joint decision-making among landowners and users on the allocation of restoration efforts.

- Carry out research on the best approaches for ensuring that individual restoration projects benefit from the landscape context and add value to it in terms of ecological and genetic connectivity, land uses and livelihood strategies. Transform the main findings into practical decision-support tools for landscape planning. Researchers should seek to consolidate the role of production systems and on-farm conservation in providing ecosystem goods and services while contributing to landscape connectivity, and should analyse the genetic impacts of different management practices and land-use patterns on tree populations.
- Advocate among politicians for policy measures and decisions in favour of landscape-scale restoration of degraded forest ecosystems.

16.6. Climate change

Survey results: *Two out of five respondents indicated that the restoration methods they use consider effects of climate change at least to some extent. At the same time, only two of the 23 respondents provided explicit approaches for anticipating climate impacts. Climate change was most commonly related to changes in species composition, with only two respondents explicitly mentioning intraspecific effects.*

Climate change is expected to change habitat and growth conditions rapidly and profoundly in most regions of the world. Some current combinations of climatic and edaphic conditions will disappear or be strongly reduced in size, while other entirely new combinations are expected to emerge. Climate change will have a strong impact on most restoration activities, yet at present few restoration practitioners appear to purposefully consider climate predictions in the design of restoration activities.

Several approaches have been suggested to build resilience to climate change in forest management and restoration initiatives. Given the uncertainty of future climatic conditions and lack of knowledge of the nature and distribution of adaptive traits of species, a prudent adaptation strategy at this time may be to increase population sizes, enhance species and genetic diversity, and ensure genetic and geographic connectivity between different biotic elements of both natural and cultural landscapes (Ledig and Kitzmiller, 1992). The underlying assumption of this approach is that high levels of genetic and species diversity and gene flow, along with large population sizes, will allow natural selection to shift fitness-related traits so that populations can adapt to changing environmental conditions (Thompson *et al.*, 2010). Larger population sizes also buffer against the risk of population extinction resulting from extreme events, such as drought, storms or fire. Many tree species exhibit a high degree of plasticity in fitness-related traits, which gives a population time to adapt to changes (O'Neill, Hamann and Wang, 2008; Thompson *et al.*, 2010; Mata, Voltas and Zas, 2012). Tree species generally also have relatively high genetic variation in adaptive traits, constituting latent adaptive potential that is expressed only when conditions change (Doi, Takahashi and Katano, 2009; Thompson *et al.*, 2010). High diversity in FRM could be combined with increased planting densities or fostering natural recruitment to increase the absolute amount of diversity in the seedlings (Ledig and Kitzmiller, 1992; Guariguata *et al.*, 2008; Chmura *et al.*, 2011) and to anticipate relatively high mortality rates. The role of generational turnover is key to the capability of tree populations to adapt through shifts in standing genetic variation, and methods to accelerate it – such as gap creation – may have to be considered.

Although many tree species have high capability for gene flow among populations (Ward *et al.*, 2005; Dick *et al.*, 2008) this may vary in accordance with species life-history traits such as pollination patterns (Vranckx *et al.*, 2012). Thus, appropriate (species-specific) conditions should be created to ensure genetic connectivity between existing and restored populations to increase the selection pool and allow better-adapted recruits (containing new genetic variation from gene flow or new combinations of existing variation) to enter the population (Newton, 2011). In a similar fashion, conditions should be created to promote the mobility and migration of plant and animal species, most notably pollinators and seed dispersers, to habitats or microhabitats within or near to restoration sites where environmental conditions best match their requirements for survival, growth and reproduction. This can be achieved, for example, by facilitating movement across hard edges such as human infrastructure (e.g. "bioducts" over highways), or by taking advantage of the topographic heterogeneity of a site being restored. Connectivity is also key in this context, particularly in terms of providing migration pathways (Rogers and Montalvo, 2004; Newton, 2011).

Degraded forest sites that require restoration typically constitute tough environments for seedling establishment and growth. When the climate simultaneously becomes harsher, natural or planted propagules experience high selection pressure. Hence, it is even more important than before to collect FRM from a large number of parent trees to maximize genetic diversity (see Chapter 2). Especially where climate change is already evident and suitable seed source populations are available, FRM should be collected from a range of environmental conditions in the same or neighbouring seed zone to increase the variability in adaptive traits and thus enhance adaptive capacity of the next generations of established tree populations. This approach relies on the assumption that the species have relatively high genetic diversity in or near the target area; it may therefore be less relevant for species whose populations have been severely reduced or for species with naturally low genetic diversity.

Additionally, the expected direction of selection could be taken into account, for example by including FRM from warmer rather than cooler environments or from drier or wetter environments, depending on general climate predictions for the region. Topographic variation in the area could be taken advantage of when sourcing more diverse FRM that may represent adaptations to varying microclimates. Furthermore, drought gradients are present at large landscape scales all over the world, and remnant trees within the more arid areas could be potential seed suppliers for restoration of the same species in neighbouring areas that are currently still more humid. Similarly, the gene pool of remaining trees on sites that are already affected by climate change can provide a useful seed source for sites with conditions that are currently less extreme but still nearing the edge of the species' tolerance. This is because such residual trees are survivors and may be better adapted to the extreme conditions. The quality of residual tree populations as seed sources, including the risk of inbreeding and the risks associated with transfer of provenance, should, however, be evaluated in all cases.

If provenance trials have been established for species of interest, it is possible to select provenances that are adapted to the expected climatic conditions of a restoration site (Ledig and Kitzmiller, 1992; O'Neill, Hamann and Wang, 2008; Wang, O'Neill and Aitken, 2010; O'Neill and Nigh, 2011). However, provenance and progeny trials that can provide knowledge about adaptive traits and climatic or environmental tolerance within species and populations exist mainly for introduced, commercially valuable species. They cover only a small proportion of species of potential interest for restoration, and most of those trials that do exist do not sample the full species' ranges. New trials should be established along gradients of relevant variables, such as elevation, latitude and aridity, both within and outside natural distribution areas of a species, and use various approaches, such as reciprocal transplanting. The main purpose of such trials would be to understand the extent and patterns of adaptive variation, rather than just choosing the best-performing provenance. Similarly, interpretation of results from existing provenance trials originally established for production purposes should be studied in the context of ecosystem restoration and adaptation.

Importantly, research is needed to understand climatic tolerance of species and their genetic variants during the critical phases of tree life cycles, such as germination and seedling establishment. If mortality risks during such phases are overlooked, the appropriateness of germplasm to given site conditions may be misjudged and the success of restoration compromised. As mentioned before, an important concern for FRM grown in nurseries is that it may be exempted from normal selection pressures during germination and initial establishment. These are precisely the stages during which trees typically experience the strongest selection pressures of their life cycle under normal field conditions. Exempting plants from such selection pressures may result in poorly adapted seeds and seedlings in future generations. Tree breeders also can contribute to enhancing survival chances of planting stock under changing environmental conditions, for example, by crossing provenances displaying adaptive traits of interest. Possible breeding strategies include: (i) selecting and breeding, for specialists, varieties that perform particularly well in specific, defined conditions, and then using them on the appropriate site types; and (ii) selecting and breeding, for generalists, varieties that perform at least moderately well in a broad range of environments (Ledig and Kitzmiller, 1992; see also Chapter 2 for other strategies). Given the uncertainties of expected climate change, breeding for generalists would be the better choice (Ledig and Kitzmiller, 1992).

In some cases habitat conditions will be altered by climate change to such an extent that the classical preferential use of local germplasm may no longer be valid. If climate conditions in the area are expected to change substantially, deliberate moving of FRM along climate gradients may need to be considered (Ledig and Kitzmiller, 1992). Examples of such strategies include assisted migration based on predictive provenancing (Chmura *et al.*, 2011; Chapter 2). Such approaches can target both provenances and species, and involve either matching of germplasm to a given restoration site or suitable sites to germplasm of interest. In mountainous areas planting stock is typically moved upwards, anticipating that future climatic conditions at higher altitudes will be similar to those presently occurring at lower altitudes. In general, FRM could be moved along latitudinal gradients to help plant communities follow changing temperature patterns. These selection strategies need to be combined with a study of the expected future environmental and climatic changes and predicted rate of change for the restoration site to ensure that FRM selected has the most potential to be resilient to future conditions. As stated previously, care must be taken to evaluate the risks of provenance transfer. Species associations such as mycorrhizal symbionts or specific pollinators should be considered when planning provenance transfer.

Spatial species distribution models can be useful tools to identify sources of FRM with potential adaptation to extreme habitat conditions and to predict the suitability of future climatic conditions for given species on particular sites (O'Neill, Hamann and Wang, 2008; Wang, O'Neill and Aitken, 2010; Sáenz-Romero *et al.*, 2010). Spatial models are already used to inform approaches to forest landscape restoration by indicating those locations within a landscape where particular restoration approaches would most likely be successful based on local environmental conditions (Newton, 2011). In a similar way, spatial models could be employed to identify areas and cases where climate change considerations in restoration may be most important, for example, at the retreating edge of a species range. The modelling approach allows projections of regeneration and spread of native forest under various anthropogenic disturbance regimes, providing insights into the potential for passive restoration approaches (Newton, 2011). However, spatial distribution models should be used with caution. Their predictions are directly related to the quality of available data on species distribution. Lack of distribution data, especially for many tropical species, and alterations to natural species distributions caused by human influence limit the application of such models. In addition, it should be noted that species distribution models are usually based on climatic factors only and often do not take into account other factors affecting species distribution, such as soil types, population genetics or species plasticity and adaptive potential (O'Neill, Hamann and Wang, 2008). Such models typically do not treat species as dynamic entities and fail to build in evolutionary processes. Hence, the potential for *in situ* adaptive change is not accounted for and predictions of changes in distribution may be over-simplified. Therefore, results of species distribution models should be groundtruthed as much as possible before using them in the design of restoration projects. In sum, distribution models can be useful for obtaining first approximations of expected species distributions or site conditions to inform restoration research, but should not be used as self-standing tools for decision-making.

As noted above, several precautions should be borne in mind with respect to moving germplasm along environmental and climatic gradients. First, there are clearly risks in the level of confidence with which predictions of future climates can be made. Even if the predictions *per se* proved to be correct, it is likely that in many cases novel environments will emerge that are currently not part of species ranges, and suitable FRM may, therefore, be difficult to identify. Second, in addition to climate, local adaptation has been shaped by numerous factors, such as photoperiod, soil conditions, the local biotic environment, and competitive and symbiotic relationships with other species and their variants, including pest and diseases (see section 16.4). Therefore, transferring germplasm on the basis of climatic gradients alone is usually not to be recommended as it risks exacerbating fitness deficits, potentially diverting attention and resources from vital efforts to bolster local population sizes through restoration and, at worst, causing genetic contamination or introducing new pests or diseases to existing populations. In spite of these precautionary observations, it should be noted that there are very different perspectives on the benefits of germplasm movement for matching expected climatic conditions, both among scientists and policy-makers (Seddon, 2010). In western Canada, for example, a forest regulation has already been changed to accommodate new seed transfer rules to better match seedlings to expected future conditions.

Finally, planting stock that is best adapted to changing environmental conditions and thus most suitable for use in restoration projects may not always be available in the country of implementation. Hence, the need for cross-border movement of germplasm will likely have to increase if ecosystem restoration projects are to be designed to respond most effectively to climate change. In light of this, there is an urgent need for countries to re-examine regulatory norms that currently impede or excessively regulate germplasm movement across political borders (Koskela *et al.*, 2010).

16.6.1. Needs for research, policy and action

- Given the uncertainty of future climate predictions, the most prudent approach to preparing for climate change for most restoration efforts is to use as much as possible of the genetic and species diversity available near the restoration site or in sites with similar (macro)environmental conditions, which will allow natural selection to take its course and move the restored population in the required direction. Restoration projects should collect forest reproductive material from a large number of parent trees and from as many sites as possible with locally varying (microenvironmental) habitat conditions. Such approaches should be used in combination with planning and management strategies explicitly designed to promote gene flow and facilitate species migration. In cases where genetic diversity is lacking and where impacts of climate change are already stressing the ecosystem, assisted migration may be necessary, taking precautions to match changing environmental conditions as closely as possible and to avoid possible associated risks to local biodiversity in target areas.

- Conduct research on the extent and distribution of plasticity and adaptive capacity in native tree species, particularly in areas that are especially vulnerable to climate change, in order to identify appropriate FRM for restoration that also maximizes resilience. Develop and test practical approaches and decision-support tools for improving ecosystem resilience through restoration. Establish provenance trials using seed sources collected from a stratified sample across the species' distribution range, on sites across environmental gradients within and beyond current species distributions. Research should also be designed to test the feasibility of assisted migration. Modelling approaches that take into account genetic diversity and selection seem a promising approach to yield timely and relevant results.

16.7. Measuring success

Evaluating the success of any action requires a clear definition of objectives or baselines against which performance can be judged. Possible objectives of ecosystem restoration are at least as diverse as its definitions, ranging from restoring tree cover, original vegetation structure and biodiversity, to ecosystem functions, services or provision of livelihoods. One of the proposed, more holistic goals for restoration is restoring ecological integrity, defined as "maintaining the diversity and quality of ecosystems, and enhancing their capacity to adapt to change and provide for the needs of future generations" (Mansourian, 2005). Another, probably more dynamic definition by lead members of the International Society of Ecological Restoration emphasizes "reinstating autogenic ecological processes by which species populations can self-organize into functional and resilient communities that adapt to changing conditions while at the same time delivering vital ecosystem services" (Alexander *et al.*, 2011b).

Despite an accumulation of experience on ecosystem restoration over the past decades, it is still quite common to measure the success of reforestation and restoration efforts mainly in terms of number of seedlings planted or their survival in the short term (Le *et al.*, 2012). Such measures ignore the importance of using good-quality FRM that is capable of establishing on the site and creating functional ecosystems over time and do not help evaluate achievement of the actual restoration objectives. If the focus on planting targets is too strong, it may divert attention from the actual objectives (the establishment of resilient plant communities) and factors critical to success, resulting in inefficient use of resources and wasting of time. Restoration success needs to be evaluated in a more holistic way, not only by restoration practitioners but also by government institutions, civil society organizations, the private sector and, importantly, funding agencies. The fact that genetic factors are still missing from the recent conceptual models and otherwise extensive lists of success indicators and drivers (Le *et al.*, 2012) is illustrative of the scale at which awareness needs to be raised about the importance of genetics in reforestation and restoration projects. Genetic variation itself is an indicator of functional and resilient ecosystems and hence also the success of restoration activities (Thompson *et al.*, 2010).

Successful re-establishment of functional ecosystems can only be truly evaluated in the long term by covering all the main stages in restoration projects (including forest establishment, growth and maturation; Le *et al.*, 2012). The problem is that such assessments extend substantially outside the time span of most restoration projects. Nevertheless, a plan or strategy for continuous monitoring of progress towards set objectives should be an integral part of any restoration effort to allow for steering and corrective management practices where necessary throughout the different stages of vegetation development. Effective monitoring requires the establishment of a baseline and a set of indicators that relate to the specific objectives of restoration. Ideally, especially for research purposes, the baseline for genetic monitoring should include the genetic structure of: (i) the remnant trees of the degraded populations in the landscape, (ii) their naturally regenerated saplings, (iii) the source populations of germplasm used, (iv) the progeny (seedlings) of this germplasm grown under nursery conditions and (v) the mating pattern in undisturbed and disturbed populations. This information would allow assessment and a better understanding of the changes in the genetic structure of species throughout the restoration process, evaluation of the genetic viability of the progeny and, eventually, assessment of the success of restoration on timescales at which fitness of species can be judged.

The genetic diversity profile of one or more healthy reference populations occurring naturally (as far as possible) in the same seed zone or ecological niche should ideally be known for assessing success in restoring genetic diversity. This could then be compared with the genetic diversity of the developing tree populations under restoration at various stages in time throughout the restoration process. Such references for target levels of genetic diversity may already exist for some species and areas, but are lacking for the large majority of species and contexts. Use of similar or standardized molecular techniques to assess diversity of restored populations would facilitate comparability and wider applicability of the findings, although this is probably not realistic for the majority of species, as techniques are constantly changing and being improved. In the long term, databases could be established containing reference levels of genetic diversity per species and for different target areas of restoration. Genetic assessments of the success of restoration projects could then be limited to measuring the genetic diversity of the restored tree populations and comparing these values with reference values from the databases. Ideally, such genetic assessments could also be extended to populations of other, naturally establishing (plant and animal) species that are representative of certain groups of species with similar functional, structural or life-history traits. In some cases it may be difficult to determine the reference target level of genetic diversity for species used in restoration activities, for example when natural populations have been nearly or completely eliminated. In such cases it may be necessary to define a baseline rather than target to allow assessment of the success of restoration activities. In addition to comparing levels of genetic diversity between restored plant populations and their natural analogues, it would also be important to assess the genetic connectivity between restored and adjacent natural, undisturbed populations.

Examples of possible indicators that could be useful for evaluating genetic composition of restored populations include genetic structure and genetic diversity for forest structure demographic characteristics, and gene flow and inbreeding for forest function (Newton, 2011). Early detection of genetic bottlenecks, which may go undetected in traditional demographic monitoring, is important to avoid potentially harmful effects and relatively easily done, for example by comparing neutral allele frequency between different generations (Luikart *et al.*, 1998; Rogers and Montalvo, 2004; Kettle, 2012). It is important to note that monitoring changes in genetic diversity must be framed in a biologically meaningful context so as to be able to interpret whether any observed changes are within a normal or desirable range, or whether they might signal some serious loss that could have negative repercussions (Rogers and Montalvo, 2004). For example, the loss of selectively neutral traits measured using molecular markers does not necessarily translate into loss of adaptive traits (Holderegger, Kamm and Gugerli, 2006). After going through an extended genetic bottleneck that dramatically reduces population size and genetic diversity, genetic variation in selectively neutral traits may require many thousands of generations to recover, whereas recovery of variation for adaptive traits may require only hundreds of generations (Milligan, Leebens-Mack and Strand, 1994; Rogers and Montalvo, 2004).

There is emerging consensus that a combination of ecological and genetic indicators would provide the best results in genetic monitoring of forested ecosystems (reviewed in Aravanopoulos, 2011). However, most restoration efforts cannot be realistically expected, at least in the short term, to include molecular studies to assess levels of genetic diversity. Moreover, most native tree species in the most biodiverse areas of the world have not been subject to molecular analyses, and most restoration practitioners are not equipped to carry out molecular analyses, even for those species that have been relatively well characterized. Two sets of indicators to evaluate genetic composition of restored tree populations are therefore needed: one for situations where molecular studies are feasible and more detailed information can be obtained, and another for situations where such studies are not feasible and information must be obtained more indirectly. Developing effective surrogate indicators for genetic diversity for wider application first requires a good understanding of various genetic, biological, ecological and management processes and how they may affect genetic diversity during restoration (Graudal *et al.*, 2013). Priority species for which to develop surrogate indicators might include those for which some baseline genetic data exist, and that are known or suspected to be particularly sensitive to human influences or environmental change (e.g. based on their life-history traits; Vranckx *et al.*, 2012; see also Jennings *et al.*, 2001; Rogers and Montalvo, 2004). Studies should be initiated in ecosystems that are simple in terms of structure and species composition to facilitate understanding of the interactions between ecological and genetic processes and consequences for genetic diversity.

Knowledge from existing studies that looked at the genetic structure, diversity and connectivity of restored tree populations in combination with ecological observations can be used to inform the development of monitoring guidelines and choice of surrogate indicators for practical restoration. The case study on *Pinus radiata* D.Don (section 15.3) illustrates how measuring genetic diversity of a target species at a degraded site, as a baseline, can be highly informative for selecting optimal approaches for restoration interventions or providing support to chosen methods. An initial assessment of genetic diversity in the seemingly highly degraded *P. radiata* population on the island of Guadalupe showed still relatively high genetic diversity and low levels of inbreeding, and led to limited intervention, simply eliminating the main factor that prevented natural regeneration (i.e. grazing by goats). Similarly, a more frequent application of genetic assessments of the success of restoration projects, such as that performed for *Banksia attenuata* (see section 15.4), would permit testing and comparison of the performance of different restoration methods for different species combinations and site contexts. Unfortunately, given the limited attention to genetic aspects in ecosystem restoration to date, little information is available on factors related to success and failure in restoring the genetic diversity and adaptive capacity of tree populations under different contexts and using different restoration methods.

While the type and amount of genetically relevant information that can be collected in practical restoration projects may (still) be limited, such efforts would be important in building a critical knowledge base on ecosystem restoration that can, in turn, support restoration research and help improve restoration guidelines. Global initiatives could be designed to collect important data from different restoration sites that could subsequently be used to better understand the genetic dimension of ecosystem restoration, conduct meta-analyses across sites, and synthesize general approaches for restoration that incorporate genetic criteria and conservation. Lessons could be learned from restoration practices across different habitats. For example, considerable effort has gone into developing standards and criteria for measuring the success of restoration of freshwater ecosystems (Palmer *et al.*, 2005) that could be relevant for restoration activities in other habitats. Among the data to be recorded at different restoration sites would be: the location of the source population of FRM; environmental description; number of trees from which FRM was collected; distances between them; amount of seed per tree; whether seed was mixed among the trees; year of seed collection and associated climatic conditions; nursery conditions; size of seedlings; whether any selection was performed in the nursery; acclimatization method prior to planting out; and whether natural regeneration also occurred on site. Enhanced collaboration between restoration practitioners and researchers would contribute to better understanding of genetic diversity and genetic processes important in restoring functional and resilient populations of native tree species, and the associated ecosystem and evolutionary processes. Lastly, while there is a dire need for better ways to synthesize and distribute knowledge from successful projects for the definition of best practices in ecosystem restoration, it is also important that failures in restoration projects are reported more systematically to help improve future strategies.

16.7.1. Needs for research, policy and action

- Conduct research for different combinations of native species, degradation states and restoration methods to understand how various biological, genetic, ecological and management processes interact and affect ecosystem functions, and the resilience of genetic diversity during restoration. Develop protocols for collecting related baseline information that are widely applicable to different species and contexts, as well as sets of genetic and surrogate indicators that allow assessment of the viability and resilience of restored tree populations.
- Through the collaboration of researchers and policy-makers, compile compelling evidence and advocate for the need to measure success in restoration projects in ways that reflect ecosystem functioning and long-term resilience. Foster collaboration between restoration researchers and practitioners to compile information, conduct meta-analyses and generalize good practices for ensuring viability of restored tree populations for functional and resilient ecosystems.

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Part 5

CONCLUSIONS AND RECOMMENDATIONS

Chapter 17

Conclusions

Evert Thomas,[1] Riina Jalonen,[1] Judy Loo,[1] Stephen Cavers,[2] Leonardo Gallo,[1,3] David Boshier,[1,4] Paul Smith,[5] Sándor Bordács[6] and Michele Bozzano[1]

[1] Bioversity International, Italy
[2] Centre for Ecology and Hydrology, Natural Environment Research Council, United Kingdom
[3] Unidad de Genética Ecológica y Mejoramiento Forestal, INTA Bariloche, Argentina
[4] Department of Plant Sciences, University of Oxford, United Kingdom
[5] Seed Conservation Department, Royal Botanic Gardens, Kew, United Kingdom
[6] Central Agricultural Office, Department of Forest and Biomass Reproductive Material, Hungary

In a world characterized by unprecedented rates of biodiversity loss, ecosystem degradation and environmental change, ecosystem restoration is more important than ever. Considerable efforts have already been made to restore degraded forested ecosystems globally, supported by experimental and anecdotal research and rapidly maturing new scientific disciplines such as restoration ecology. However, there is a need to further upscale and mainstream activities. Policy-makers are increasingly recognizing the potential of ecosystem restoration for mitigating and reversing a wide range of environmental problems and associated opportunities for socioeconomic benefits. The most important future challenge will be to translate the knowledge generated from research into widespread sustainable practice. It is imperative that restoration practice develops strong multidisciplinary approaches that include a stronger focus on important but previously neglected factors. The genetic composition (diversity and adaptedness) of tree populations in restored ecosystems is often overlooked despite its fundamental importance for the success of restoration in both the short and the long term. The objective of this thematic study is to highlight the breadth and depth of genetic aspects that need to be considered in ecosystem restoration using native tree species, and to propose recommendations for researchers, policy-makers and restoration practitioners to better address the deficiencies that may currently compromise the success of some restoration efforts.

There is a dire need to develop elemental, practical and convincing recommendations and guidelines for selecting, collecting and propagating genetically diverse and appropriately adapted planting material that is specifically tailored to ecosystem restoration. General guidelines for selecting and collecting planting material for restoration are largely compatible with those traditionally used in forestry and agroforestry and should build on them. However, it is clear that some issues need more emphasis in restoration. In particular, the traditional rule of thumb that local seed is generally the best choice when sourcing planting material for restoration may no longer be universally applicable. Although local seeds may still be the preferred option in the absence of knowledge about the qualities of germplasm sources, scientific evidence is increasingly showing that local tree populations may not be adapted to environmental conditions that are already degraded or those that are expected in the future. Even if local populations are adapted to future conditions, in many areas they may already be too degraded or fragmented to constitute good sources of seed for restoring viable and resilient self-sustaining populations. Such fragments may no longer have the adaptive capacity required to cope with environmental changes, although responses may be species-specific. More research and decision-support tools are needed to help evaluate the quality of local germplasm as seed sources. In some cases it may be appropriate to source seed from more remote, larger populations growing under ecological conditions similar to the planting site, but perhaps with climatic conditions more similar to those expected in the near future. However, special care should be taken to avoid potentially harmful effects from genetic contamination of remaining resident populations.

Given the uncertainty of predictions of future climate and the limitations of current knowledge about the vast majority of native tree species, for most restoration efforts the most prudent approach for anticipating the impacts of environmental change would be to maximize genetic and species diversity that is well-matched to the target sites, while at the same time creating favourable conditions for connectivity (i.e. for gene flow and species migration) and regeneration. There is still considerable controversy about the necessity, utility, feasibility and risk of deliberately moving germplasm over long distances to help tree species and populations track changing environmental conditions. The most informative and safest way to tackle this uncertainty would be to expand the use of provenance trials to cover a wider range of environments, more native species and a wider range of traits (e.g. the consequences of germplasm transfer for associated organisms are rarely considered). There is considerable potential for learning about the performance and adaptive potential of propagation material from known sources in already established and planned restoration sites. In addition, increased use of ecological zones or other proxies for adaptive variation, such as environmental gradients, would help restoration practitioners chose the best sources of site-matched planting material.

Another major constraint on diversification of restoration efforts and optimization of different approaches to local conditions is the availability of adequate planting material. At present, the choice of species and propagation material of many, if not most, restoration projects is heavily influenced by what is available in (commercial) nurseries or that can be easily collected. This may lead to a mismatch between planting stock and the conditions at the restoration site, or may even result in the selection of suboptimal restoration approaches. First and foremost, adequate political incentives and public policies are key for motivating nurseries, especially commercial operations, to widen the choice of planting material. A second priority is to improve restoration practitioners' and nursery managers' access to existing and new knowledge about the genetics, biology and management of native species. This implies the development of better tools and communication strategies to improve information sharing. In particular, knowledge has to move beyond the scientific restoration community and academic journals to reach the much broader community of restoration practitioners, nursery managers and seed suppliers. This may require translation of information into different languages and accessible styles, increased efforts to raise awareness and strengthening of capacity specific to the cultural, socioeconomic and gender context of the target audience.

A third priority for widening the choice of propagation materials is to improve communication channels, cooperation and feedback between seed suppliers, nurseries and restoration projects. During the planning stages, restoration practitioners should inform nursery managers or seed suppliers about the planting material they want to use and help them to identify potential seed sources. Restoration practitioners, if not linked to the nursery where their planting material is produced, should provide feedback to nursery managers about problems experienced with their planting material in the field. To compensate for the limited choice of planting material in existing (commercial) nurseries or seed banks, or to complement the germplasm that is available, restoration projects could set up their own (project or community) nurseries or seed banks to produce the desired plant material. In either case, adequate time needs to be reserved for sourcing the germplasm, including identifying source populations and flowering and fruiting seasons of the different species.

Record-keeping is essential for good genetic management in individual restoration projects, as well as at nurseries and seed banks. Adoption of a certification scheme such as the OECD Forest Seed and Plant Scheme would ensure not only systematic record-keeping, but also traceability of germplasm movement. Looking to the future, records of the genetic base and source of materials used in restoration projects will also help to evaluate the qualities germplasm and inform decisions about where to make future collections of planting material in restored vegetation sites.

Political commitment and supportive regulatory frameworks can help promote demand for and supply of good-quality germplasm of native tree species. Governments must support and guide restoration initiatives through carefully defined policies and financial support. Currently, few countries have regulatory frameworks and funding schemes for ecosystem restoration in place. Strengthening national capacities to restore genetically diverse, healthy and resilient forested ecosystems is also important and should be supported by integrating genetic aspects into curricula on biodiversity conservation and restoration, and the provision of appropriate training to restoration practitioners.

The actions proposed above can be expected to contribute to increased popularity and use of native tree species in restoration projects. Nonetheless, the use of exotics may sometimes be justified, particularly as nurse plants to facilitate subsequent establishment of native species by improving (micro)environmental conditions in severely degraded environments, or when native species with comparable properties have not yet been identified. However, a preference for exotics even when native alternatives are available suggests that there are other factors that constrain the use of natives besides availability and knowledge of their biology. More research is, therefore, necessary to understand underlying factors that may affect species selection and the trade-offs related to the use of exotic versus native species in different contexts, such as behavioural patterns or local perceptions of the ecological and socio-economic value of species.

In addition to their contribution to mitigating and countering ecosystem degradation, restoration projects also hold great potential for contributing to biodiversity conservation. Restoration practitioners should plan to integrate their activities in the wider landscape from the outset, and consider how they can both benefit from the landscape context and in turn bolster biodiversity conservation. A stronger focus on species associations, such as microbial symbionts and pollinators, can enhance the chances of survival of planted or recruited trees, optimize the use of available resources on site, and increase the resilience of the plant community to biotic and abiotic stresses. Genetic connectivity contributes to the maintenance of population-level diversity through pollen flow, which reduces the likelihood of inbreeding and results in new genetic combinations that may be better adapted to changing environmental conditions. Genetic connectivity also facilitates migration of populations to more suitable habitats through seed dispersal. On the other hand, restoration efforts have considerable potential to support conservation of threatened or endangered native tree species, their genetic diversity and associated species, for example, by restoring and maintaining genetic diversity of species across a range of ecological conditions and establishing genetically diverse sources of propagation material.

New approaches and tools are needed to enable communication and coordination among restoration practitioners in order to realize the biodiversity potential of restoration projects. Policy-makers can play a facilitating role in bringing together stakeholders involved in landscape planning. This could lead to development of practical guidelines on how to best organize ecological and genetic connectivity and conservation at landscape level, which could serve as policy support tools. Lessons can be learned from reconstructing the historical ecologies of particular target areas, which could guide the design of connectivity across different landscape scenarios. In this context, clarity of the objectives and species for which connectivity is sought is key.

Finally, in monitoring and assessing the success of restoration it is important that restoration practitioners systematically integrate genetic aspects in ways that reflect ecosystem functionality and resilience in the long term. Reference levels against which changes in genetic diversity and structure of plant populations can be assessed include baseline genetic diversity at the start and the target genetic diversity (e.g. by comparison with known natural populations). There is an urgent need to develop indicators that allow measurement of the success of restoration efforts to (re-)establish self-sustaining plant communities, including the likelihood of long-term population viability of the tree species. Such indicators may require direct monitoring of genetic parameters in restoration sites or monitoring of other (less-costly) variables as proxies for genetic diversity. Ideally, indicators should be few and designed in such a fashion that they are based on ecological or biological measures and can be widely applied across restoration projects. In general, restoration interventions should incorporate a monitoring (and evaluation) strategy that extends beyond the establishment of seedlings and, ideally, continues long enough to assess the reproductive success of species in restored ecosystems. Comparative research should be carried out to analyse the success of various restoration approaches, including – critically – recording and reviewing failures as well as promoting methods that have successfully restored viable populations across varying states of degradation and landscapes. This will allow identification of good practices and potential problems associated with genetic quality under various restoration approaches and contexts, and facilitate formulation of solutions to overcome them, thus contributing to more successful (ecological and socioeconomic) restoration of forested ecosystems.

17.1. Recommendations arising from the thematic study

17.1.1. Recommendations for research

- Evaluate the impact of different restoration methods on the genetic diversity of restored tree populations.
- Expand knowledge on native species, particularly with respect to their ecological and livelihood importance, propagation methods and genetic variability, and identify ways to overcome constraints that limit their use in restoration.
- Develop, make available and support the adoption of decision-support tools for: (i) collecting and propagating germplasm in a way that ensures a broad genetic base of restored tree populations; (ii) matching of species and provenances to restoration sites based on (current and future) site conditions, predicted or known patterns of variation in adaptive traits, and availability of seed sources; and (iii) landscape-level planning in restoration projects.
- Develop protocols and practical indicators to monitor and evaluate the genetic diversity of tree populations in restoration efforts as an indicator of the viability and resilience of ecosystems.
- Intensify research on the ecology of mycorrhizal and bacterial symbiotic systems, focusing on the most commonly used tree species and their symbiotic partners to increase the resilience of plant associations in restoration against biotic and abiotic stresses.

17.1.2. Recommendations for restoration practice

- Give priority to the use of native tree species in restoration projects.
- Strive to use propagation material that is well matched to the environmental

conditions of the restoration site and represents a broad genetic base.
- Given the uncertainty of predictions of future climate, aim to promote resilience by maximizing species and genetic diversity from sources that are similar to the site conditions, encouraging gene flow and generational turnover, and facilitating species migration to allow natural selection to take place.
- Plan for the sourcing of propagation material of desired species and associated information well before the intended planting or seeding time to ensure that optimal material for the site and restoration objectives can be identified and produced.
- Consistently plan restoration efforts in the landscape context and seek to integrate them into the surrounding landscape matrix.

17.1.3. Recommendations for policy

- Create an enabling national policy environment that fosters long-term, ecologically based forest management that explicitly favours the use of native species in ecosystem restoration and genetic conservation and provides adequate financial support.
- Put in place supportive regulatory frameworks that guide the production and supply of propagation material of native tree species and the use of adequately diverse material of appropriate origin in restoration efforts.
- Broaden education and training curricula to promote understanding of the importance of using native species and genetically diverse and appropriate propagation material, as well as appropriate approaches, in restoration projects.

There is renewed interest in the use of native tree species in ecosystem restoration for their biodiversity benefits. Growing native tree species in production systems (e.g. plantation forests and subsistence agriculture) can also ensure landscape functionality and support for human livelihoods.

Achieving full benefits, however, requires consideration of genetic aspects that are often neglected, such as suitability of germplasm to the site, quality and quantity of the genetic pool used and regeneration potential. Understanding the extent and nature of gene flow across fragmented agro-ecosystems is also crucial to successful ecosystem restoration.

This study, prepared within the ambit of *The State of the World's Forest Genetic Resources*, reviews the role of genetic considerations in a wide range of ecosystem restoration activities involving trees. It evaluates how different approaches take, or could take, genetic aspects into account, thereby leading to the identification and selection of the most appropriate methods.

The publication includes a review and syntheses of experience and results; an analysis of successes and failures in various systems; and definitions of best practices including genetic aspects. It also identifies knowledge gaps and needs for further research and development efforts. Its findings, drawn from a range of approaches, help to clarify the role of genetic diversity and will contribute to future developments.

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